WHITE PAPER

Flow Control Structures

Prepared for

Washington Department of Fish and Wildlife

December 2007 (Working Draft – Do Not Cite)

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Prepared for

Prepared for
Washington Department of Fish and Wildlife
600 Capitol Way North
Olympia, Washington
98501-1091

Prepared by

Herrera Environmental Consultants, Inc.
2200 Sixth Avenue, Suite 1100
Seattle, Washington 98121
Telephone: 206/441-9080

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Executive Summary

The Revised Code of Washington (RCW) directs the Washington Department of Fish and Wildlife (WDFW) to “preserve, protect, perpetuate, and manage” the fish and wildlife species of the state as its paramount responsibility (RCW 77.04.012). Under RCW 77.55, any construction or work that uses, diverts, obstructs, or changes the natural bed or flow of state waters requires a Hydraulic Project Approval (HPA) issued by WDFW. The purpose of the HPA program is to ensure that hydraulic projects are completed in a manner that prevents damage to public fish and shellfish resources and their habitats. To ensure that the HPA program complies with the Endangered Species Act (ESA), WDFW is developing a programmatic multispecies Habitat Conservation Plan (HCP) to obtain an Incidental Take Permit (ITP) from the U.S. Fish and Wildlife Service (USFWS) and the National Oceanic and Atmospheric Administration (NOAA) Fisheries Service (also known as NOAA Fisheries), in accordance with Section 10 of the ESA. For WDFW, the objective is to avoid and/or minimize the incidental take of those aquatic species potentially considered for coverage under the HCP (referred to in this white paper as “HCP species”) resulting from activities conducted under an HPA.

The HCP will address the impacts, potential for take, and mitigation measures for effects on HCP species from hydraulic projects that require HPAs. WDFW’s intent is to build the scientific foundation for the effort to prepare an HCP for hydraulic projects that receive HPAs. To accomplish this, WDFW is compiling the best available scientific information related to the impacts, potential for incidental “take” of species that may be covered in the HCP (as defined in the ESA; see Section 9 [*Potential Rick of Take]* of this white paper for a definition of “take”), and possible management directives and mitigation measures to avoid and/or minimize potential take to the maximum extent practicable. As the HPA authority covers all waters of the state, this white paper considers hydraulic project impacts in both freshwater and marine environments.

This white paper is one of a suite of white papers prepared to establish the scientific basis for the HCP and to assist WDFW decision-making on what specific HPA activities should be covered by the HCP. This particular white paper compiles and synthesizes existing scientific information on flow control structures, which in this white paper analysis include dams, weirs, dikes and levees, outfalls, intakes and diversions, and tide gates.

The objectives of this white paper are to:

- Compile and synthesize the best available scientific information related to the potential human impacts on HCP species, their habitats, and associated ecological processes resulting from the construction, maintenance, repair, replacement, modification, operation, and removal (hereafter collectively referred to as construction and maintenance) of dams, weirs, dikes and levees, outfalls, intakes and diversions, and tide gates.

- Use this scientific information to estimate the circumstances, mechanisms, and risks of incidental take potentially or likely to result from the construction and repair of flow control structures.

- Identify appropriate and practicable measures, including policy directives, conservation measures, and best management practices (BMPs), to avoid, minimize, or mitigate the risk of incidental take of HCP species.

The literature review conducted for this white paper identified six impact mechanisms that could potentially affect HCP species. These mechanisms of impact are both direct and indirect and can have temporary, short-term effects or permanent, long-term effects. The impact mechanisms analyzed in this white paper are:

- Construction and maintenance activities
- Hydraulic and geomorphic modifications
- Ecosystem fragmentation
- Riparian vegetation modifications
- Aquatic vegetation modifications
- Water quality modifications.

The white paper includes discussion of the potential direct and indirect impacts on the 52 HCP species and their habitats due to exposure to the six identified impact mechanisms. Following this discussion, an evaluation of potential for take of the 52 HCP species is included based on a separate analysis conducted using exposure-response matrices for each of the HCP species. This white paper also reviews data gaps and uncertainties and estimates the risk of take. In addition, habitat protection, conservation, mitigation, and management strategies that could avoid, minimize, or mitigate the identified potential impacts are provided. Key elements of the white paper are:

- Identify the distribution of the 52 HCP species (i.e., whether they use fresh water, marine water, or both) and their habitat requirements.

- Identify the risk of "take" associated with each of these impact mechanisms based on the distribution information.

- Identify cumulative impacts.

- Identify data gaps.

- Identify habitat protection, conservation, and mitigation strategies.

1.0 Introduction

The Revised Code of Washington (RCW) directs the Washington Department of Fish and Wildlife (WDFW) to "preserve, protect, perpetuate, and manage" the fish and wildlife species of the state as its paramount responsibility (RCW 77.04.012). Under RCW 77.55, any construction or work that uses, diverts, obstructs, or changes the natural bed or flow of state waters requires a Hydraulic Project Approval (HPA) issued by WDFW. The purpose of the HPA program is to ensure that these activities are completed in a manner that prevents damage to public fish and shellfish resources and their habitats. To ensure that the HPA program complies with the Endangered Species Act (ESA), WDFW is developing a programmatic multispecies Habitat Conservation Plan (HCP) to obtain an Incidental Take Permit (ITP), in accordance with Section 10 of the ESA, from the U.S. Fish and Wildlife Service (USFWS) and the National Oceanic and Atmospheric Administration (NOAA) Fisheries Service (also known as NOAA Fisheries). For WDFW, the benefits of an HCP are to contribute to the long-term conservation of both listed and unlisted species through the minimization and mitigation of impacts on those species and their habitats, while ensuring that WDFW can legally proceed with the issuance of HPAs that might otherwise result in the incidental "take" of ESA-listed species (as defined in the ESA; see Section 9 [*Potential Rick of Take]* of this white paper for a definition of "take").

The HCP will identify the impacts on those aquatic species considered for coverage under the HCP, the potential for take, and mitigation measures for hydraulic projects that require HPAs. This white paper is part of the effort to compile the best available scientific information to protect these species during the construction, maintenance, repair, and operation of flow control structures. To accomplish this, WDFW is identifying management directives and mitigation measures to avoid and/or minimize potential take to the maximum extent practicable. Because the HPA authority covers all waters of the state, this white paper considers hydraulic project impacts in both freshwater and marine environments. This white paper is one of a suite of white papers being prepared to establish the scientific basis for the HCP and to assist WDFW decision-making regarding what specific HPA activities should be covered by the HCP and what minimization and mitigation measures can be implemented to address the potential effects of hydraulic projects. This white paper addresses impacts and mitigation/minimization measures to be applied to the construction and maintenance of flow control structures. Species considered for coverage under the HCP (referred to in this white paper as "HCP species") are listed in Table 1-1. For the purpose of this white paper, some of the HCP species have been grouped where appropriate (and each group is separated by a gray-shaded line in Table 1-1).

Table 1-1. The 52 HCP species addressed in this white paper.

Common Name	Scientific Name	Status[a]	Habitat
Chinook salmon	*Oncorhynchus tshawytscha*	FE/FT/SC	Freshwater, Estuarine, Marine
Coho salmon	*Oncorhynchus kisutch*	FT/FSC	Freshwater, Estuarine, Marine
Chum salmon	*Oncorhynchus keta*	FT/SC	Freshwater, Estuarine, Marine
Pink salmon	*Oncorhynchus gorbuscha*	SPHS	Freshwater, Estuarine, Marine
Sockeye salmon	*Oncorhynchus nerka*	FE/FT/SC	Freshwater, Estuarine, Marine
Steelhead	*Oncorhynchus mykiss*	FE/FT/SC	Freshwater, Estuarine, Marine
Coastal cutthroat trout	*Oncorhynchus clarki clarki*	FSC	Freshwater, Estuarine, Marine
Redband trout	*Oncorhynchus mykiss*	FSC	Freshwater
Westslope cutthroat trout	*Oncorhynchus clarki lewisii*	FSC	Freshwater
Bull trout	*Salvelinus confluentus*	FT/SC	Freshwater, Estuarine
Dolly Varden	*Salvelinus malma*	FP	Freshwater, Estuarine
Pygmy whitefish	*Prosopiim coulteri*	FSC/SS	Freshwater
Olympic mudminnow	*Novumbra hubbsi*	SS	Freshwater
Lake chub	*Couesius plumbeus*	SC	Freshwater
Leopard dace	*Rhinichthys falcatus*	SC	Freshwater
Margined sculpin	*Cottus marginatus*	FSC/SS	Freshwater
Mountain sucker	*Catostomus platyrynchus*	SC	Freshwater
Umatilla dace	*Rhinichthys umatilla*	SC	Freshwater
Pacific lamprey	*Lampetra tridentata*	FSC	Freshwater, Estuarine, Marine
River lamprey	*Lampetra ayresi*	FSC/SC	Freshwater, Estuarine
Western brook lamprey	*Lampetra richardsoni*	FSC	Freshwater
Green sturgeon	*Acipenser medirostris*	FSC/FT/SPHS	Freshwater, Estuarine, Marine
White sturgeon	*Acipenser transmontanus*	SPHS	Freshwater, Estuarine, Marine
Longfin smelt	*Spirinchus thaleichthys*	SPHS	Freshwater, Estuarine, Marine
Eulachon	*Thaleichthys pacificus*	FC/SC	Freshwater, Estuarine, Marine
Pacific sand lance	*Ammodytes hexapterus*	SPHS	Marine & Estuarine
Surf smelt	*Hypomesus pretiosus*	SPHS	Marine & Estuarine
Pacific herring	*Clupea harengus pallasi*	FC/SC	Marine & Estuarine
Lingcod	*Ophiodon elongatus*	SPHS	Marine & Estuarine
Pacific cod	*Gadus macrocephalus*	FSC/SC	Marine (occ. Estuarine)
Pacific hake	*Merluccius productus*	FSC/SC	Marine & Estuarine
Walleye pollock	*Theragra chalcogramma*	FSC/SC	Marine (occ. Estuarine)
Black rockfish	*Sebastes melanops*	SC	Marine & Estuarine
Bocaccio rockfish	*Sebastes paucispinis*	SC	Marine & Estuarine

Table 1-1 (continued). The 52 HCP species addressed in this white paper.

Common Name	Scientific Name	Status[a]	Habitat
Brown rockfish	*Sebastes auriculatus*	SC	Marine & Estuarine
Canary rockfish	*Sebastes pinniger*	SC	Marine & Estuarine
China rockfish	*Sebastes nebulosis*	SC	Marine & Estuarine
Copper rockfish	*Sebastes caurinus*	FSC/SC	Marine & Estuarine
Greenstriped rockfish	*Sebastes elongates*	SC	Marine & Estuarine
Quillback rockfish	*Sebastes maliger*	FSC/SC	Marine & Estuarine
Redstripe rockfish	*Sebastes proriger*	SC	Marine & Estuarine
Tiger rockfish	*Sebastes nigrocinctus*	SC	Marine & Estuarine
Widow rockfish	*Sebastes entomelas*	SC	Marine & Estuarine
Yelloweye rockfish	*Sebastes ruberrimus*	SC	Marine & Estuarine
Yellowtail rockfish	*Sebastes flavidus*	SC	Marine & Estuarine
Olympia oyster	*Ostrea lurida*	SPHS	Marine & Estuarine
Northern abalone	*Haliotis kamtschatkana*	FSC/SC	Marine
Newcomb's littorine snail	*Algamorda subrotundata*	FSC/SC	Marine
Giant Columbia River limpet	*Fisherola nuttalli*	SC	Freshwater
Great Columbia River spire snail	*Fluminicola columbiana*	FSC/SC	Freshwater
California floater (mussel)	*Anodonta californiensis*	FSC/SC	Freshwater
Western ridged mussel	*Gonidea angulata*	None	Freshwater

Notes: For the purpose of this white paper, some of the HCP species have been grouped when appropriate (each group is separated by a gray-shaded line).

[a] Status:

FE=Federal Endangered	FSC = Federal Species of Concern
FP=Federal Proposed	SC = State Candidate
FT = Federal Threatened	SS = State Sensitive
FC = Federal Candidate	SPHS = State Priority Habitat Species

2.0 Objectives

The objectives of this white paper are to:

- Compile and synthesize the best available scientific information related to the potential human impacts on HCP species, their habitats, and associated ecological processes resulting from the creation, construction, maintenance, installation, repair, replacement, modification, and removal (hereafter collectively referred to as construction and maintenance) of flow control structures.
- Use this scientific information to estimate the circumstances, mechanisms, and risks of incidental take potentially or likely resulting from the construction and maintenance of flow control structures.
- Identify appropriate and practicable measures, including policy directives, conservation measures, and best management practices (BMPs), to avoid and/or minimize the risks of incidental take of HCP species.

3.0 Methods

Information presented in this white paper is based primarily on the compilation and synthesis of the best available scientific information related to human impacts on HCP species, their habitats, and associated ecological processes. The methods used here included the acquisition of existing literature, followed by an analysis of impacts based on a review of the literature. The conceptual framework for assessing potential impacts is described in detail in Section 6, and below is a discussion of the literature acquisition and review process.

To acquire literature supporting the best available scientific information, an extensive search of the available literature was conducted using the Thomson Scientific Web of Science (Thomson Scientific Web of Science 2007), which has electronic access to more than 8,500 scientific journals encompassing all fields of environmental science. This yielded several hundred relevant publications, most published within the last 10 years. In addition, literature cited in previous white papers and conference proceedings from the last four Puget Sound–Georgia Basin Research Conferences was reviewed to identify relevant "gray literature" sources. The University of Washington School of Aquatic and Fisheries Sciences, Fisheries Research Institute Reports (UW-FRI) database was also searched (this database includes more than 500 reports pertaining to research conducted by Fisheries Research Institute personnel from its inception to the present). A thorough search of theses in the Summit system of libraries was performed to locate relevant student work. (Summit is a library catalog that combines information from Pacific Northwest academic libraries, including the Orbis and Cascade systems, into a single database available at URL = http://summit.orbiscascade.org/.). Finally, because this white paper was prepared by a diverse group of scientists from a wide range of backgrounds, many other primary resources (e.g., consultant reports and textbooks) were found in the personal collections of staff with Herrera (the consulting firm working with WDFW to prepare this white paper).

To obtain as much relevant species-specific information as possible, a literature review using Thompson Scientific Web of Science was conducted to collect information related to the individual stressors for the 52 HCP species. A keyword search of the scientific name and/or common name for each species in Table 1-1 was conducted. For those species where the search returned more than 1,000 references, a few recent citations were selected for inclusion. Species in this category were the five salmon species (sockeye, chum, pink, coho, and Chinook), steelhead, and coastal cutthroat trout. For the remaining species, every reference in the search result was reviewed for the relevance of species-specific information to be included in this white paper. For several species, searches for scientific names and common names returned no references. These species included the margined sculpin, giant Columbia River limpet, great Columbia spire snail, western ridged mussel, river lamprey, longfin smelt, Newcomb's littorine snail, and many of the rockfish species.

To identify data gaps and evaluate the state of scientific knowledge applicable to the potential impacts of flow control structures on the HCP species and their habitats, the acquired literature was examined to assess the broader issue of how these species use aquatic habitats and how flow control structures and their construction may alter habitat functions.

Existing literature reviews, peer-reviewed journal articles, books, theses/dissertations, and technical reports were reviewed for information specific to aquatic species and their interaction with each flow control structure subactivity type. Through this process, a collection of information was assembled on the life history, habitat uses, and the potential impacts that these structures pose to HCP species.

Reference material from each of the above databases was compiled in an Endnote personal reference database (i.e., Endnote version X). Reference types collected and entered into the database included journal articles, reports, web pages, conference proceedings, theses, statutes, books, and book sections. Each entry in the database included descriptive information, including author(s), year, title, volume, pages, and publisher. Whenever an electronic copy of the reference material was available, a link between the reference entry and a PDF copy of the reference material was included in the database. If an electronic (.PDF) copy of a reference was not available, a hardcopy of the material was kept on file. All reference materials cited in the literature review were either linked to the reference database or retained in an associated file as a hardcopy.

Endnote X is the industry standard software for organizing bibliographic information. It features a fully searchable and field sortable database that can contain an unlimited number of references. Reference information is entered into the database either by direct import from online databases or by manually entering the reference information into reference type templates. Once all the references were entered, the database was used for organizational and archival purposes. The final database is included as an electronic appendix to this white paper (Appendix B).

4.0 Hydraulic Project Description

RCW 77.55.011(7) defines a hydraulic project as “the construction or performance of work that will use, divert, obstruct, or change the natural flow or bed of any of the salt or freshwaters of the state.” Flow control structures considered in this white paper include: dams, weirs, dikes and levees, outfalls, intakes and diversions, and tide gates (referred to here as “subactivity types”).

The mechanisms of impact on HCP species associated with these projects include the impacts from the ongoing presence of these structures, which in most cases is of a permanent nature, as well as impacts from the related construction, maintenance, repair, and removal (including demolition) activities that could result in modifications to riverine, lacustrine, and marine processes. These impacts include modifications to hydraulic and geomorphic characteristics, aquatic and riparian vegetation, and water quality that could result in direct and indirect effects on HCP species. They also include the effects of fish handling, relocation, and exclusion that may be associated with such activities. This white paper does not, however, address fish passage and fish screening. For information on those activities, see their respective white papers (Herrera 2007a, 2007b).

For the purposes of this white paper:

- **Dams** are structures built within a stream to control flow for flood control, divert flow for irrigation, or to utilize flow for generation of hydropower.

- **Weirs** are structures that can partially or fully span the channel for purposes of flow control and water diversion. (Weir-type structures used for fish passage management and habitat modification are addressed in other white papers [Herrera 2007a, 2007e, respectively]).

- **Dikes and levees** are built to maintain flows within a confined channel for flood control purposes, or are used to convert estuarine habitat into agricultural fields or freshwater habitat (e.g., used on WDFW lands and federal wildlife refuges to provide waterfowl habitat/hunting areas).

- **Outfalls** are utilized to move water from one place to another, typically another body of water. They may convey irrigation water, stormwater, or other waste materials.

- **Water intakes and diversion structures** are utilized to divert water from a stream to another place for irrigation or other purposes (e.g., creation of fish habitat), or to maintain water in an existing or new channel for flood control. Intakes are generally utilized for the removal of water from a water body for irrigation, domestic use, or stock watering.

- **Tide/flood gates** are typically utilized for preventing saltwater intrusion or for flood control purposes in nontidal areas. They allow water to flow downstream to a marine or estuarine area while not allowing tidal water to backflow upstream. In agricultural areas, they prohibit salt water from entering croplands.

This white paper also summarizes the provisions in the Washington Administrative Code (WAC 220-110) that apply to flow control projects in both fresh and saltwater environments. These are presented in Table 4-1 for each subactivity type. Table 4-1 also identifies activities that may be related to the topics covered in this white paper, but that are not necessarily an implicit component of the activity as specified in the WAC.

Table 4-1. WAC sections potentially applicable to flow control structures.

Subactivity Types	Freshwater WACs (direct & indirect applicability)	Saltwater WACs (direct & indirect applicability)
Dams	220-110-050 (freshwater banks)* 220-110-120 (temporary bypass) 220-110-130 (dredging)* 220-110-140 (gravel removal)* 220-110-150 (LWD) 220-110-190 (diversions) 223-110-223 (lake bank)*	220-110-250 (habitats of concern) 220-110-270 (common) 220-110-271 (prohibited work windows) 220-110-280 (non-SFRM bank)* 220-110-320 (dredging)*
Weirs	220-110-050 (bank)* 220-110-080 (channel change) 220-110-120 (temporary bypass) 220-110-130 (dredging)* 220-110-140 (gravel removal)* 220-110-150 (LWD) 220-110-223 (lake bank)*	220-110-250 (habitats of concern) 220-110-270 (common) 220-110-271 (prohibited work windows) 220-110-280 (non-SFRM bank)* 220-110-285 (SFRM bank)* 220-110-320 (dredging)*
Dikes and levees	220-110-050 (freshwater banks) 220-110-120 (temporary bypass) 220-110-130 (dredging)* 220-110-140 (gravel removal)* 220-110-150 (LWD) 223-110-223 (lake bank)	220-110-250 (habitats of concern) 220-110-270 (common) 220-110-271 (prohibited work windows) 220-110-280 (non-SFRM bank) 220-110-320 (dredging)*
Outfalls	220-110-050 (freshwater banks)* 220-110-100 (conduit crossings) 220-110-120 (temporary bypass) 220-110-130 (dredging)* 220-110-140 (gravel removal)* 220-110-150 (LWD) 220-110-170 (outfalls) 220-110-223 (lake bank)*	220-110-250 (habitats of concern) 220-110-270 (common) 220-110-271 (prohibited work windows) 220-110-280 (non-SFRM bank)* 220-110-310 (utility lines) 220-110-320 (dredging)*
Intakes and diversions	220-110-050 (freshwater banks)* 220-110-120 (temporary bypass) 220-110-130 (dredging)* 220-110-140 (gravel removal)* 220-110-150 (LWD) 220-110-190 (diversions) 223-110-223 (lake bank)*	220-110-250 (habitats of concern) 220-110-270 (common) 220-110-271 (prohibited work windows) 220-110-280 (non-SFRM bank)* 220-110-320 (dredging)*
Tide gate	220-110-050 (freshwater banks)* 220-110-070 (water crossing structures)* 220-110-100 (conduit crossings)* 220-110-120 (temporary bypass) 220-110-130 (dredging)* 220-110-140 (gravel removal)* 220-110-150 (LWD) 220-110-170 (outfalls)* 220-110-190 (diversions) 220-110-223 (lake bank)*	220-110-250 (habitats of concern) 220-110-270 (common) 220-110-271 (prohibited work windows) 220-110-280 (non-SFRM bank)* 220-110-310 (utility lines)* 220-110-320 (dredging)*

Note: * indicates that the activity may be related to the topics covered in this white paper, but it is not necessarily an implicit component of the activity as specified in the WAC. LWD = large woody debris; SFRM = single-family residential marine.

5.0 Potentially Covered Species and Habitat Use

This white paper identifies what is known about activities associated with the construction and maintenance of dams, weirs, dikes and levees, outfalls, intakes and diversions, and tide gates and how each can pose a risk of take for the 52 HCP species. To understand species-specific impacts, it is important to understand the geographic distribution and habitat use of each species. Table 5-1 lists the scientific name, Water Resource Inventory Area (WRIA), and Tidal Reference Area of occurrence, as well as the reproductive patterns and habitat requirements of each HCP species. Through the identification of species-specific habitat needs, the risk of take associated with the construction and maintenance of flow control structures can be identified. Once the potential for take is identified, it can then be avoided. If unavoidable, the risk of take can be minimized by design and/or through the use of conservation and protection measures. (See Section 9 [*Potential Rick of Take*] and the exposure-response matrices for each of these species presented in Appendix A.)

Table 5-1. Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Chinook salmon	*Oncorhynchus tshawytscha*	01–42, 44–50	All	**General Information (Habitats and Feeding/Life-history Types)** NOAA Fisheries recognizes eight evolutionarily significant units (ESUs) of Chinook salmon in Washington: (1) Upper Columbia River spring-run; (2) Snake River spring/summer run; (3) Snake River fall-run; (4) Puget Sound; (5) lower Columbia River; (6) Washington coast; (7) Mid-Columbia River spring-run; and (8) Upper Columbia River summer/fall-run. Chinook salmon exhibit one of two life-history types, or races: the stream-type and the ocean-type. Stream-type Chinook tend to spend 1 (or less frequently 2) years in freshwater environments as juveniles prior to migrating to salt water as smolts. Stream-type Chinook are much more dependent on freshwater stream ecosystems than ocean-type Chinook. Stream-type Chinook do not extensively rear in estuarine and marine nearshore environments; rather, they head offshore and begin their seaward migrations. Ocean-type Chinook enter salt water at one of three phases: immediate fry migration soon after yolk is absorbed, fry migration 60–150 days after emergence, and fingerling migrants that migrate in the late summer or fall of their first year. Ocean-type Chinook are highly dependent on estuarine habitats to complete their life history. Chinook generally feed on invertebrates but become more piscivorous with age. **Reproduction/Life History** Chinook runs are designated on the basis of adult migration timing: • Spring-run Chinook: Tend to enter fresh water as immature fish, migrate far upriver, and finally spawn in the late summer and early autumn. • Fall-run Chinook: Enter fresh water at an advanced stage of maturity, move rapidly to their spawning areas on the mainstem or lower tributaries of the rivers, and spawn within a few days or weeks of freshwater entry. • Spring Chinook: Spawning occurs from mid-July to mid-December, and incubation lasts approximately 1.5–7 months, depending on temperature. Emergence follows, 6–8 months from fertilization. • Fall Chinook: Spawning occurs from late October to early December, with incubation occurring for 1–6 months. Emergence follows, approximately 6 months after fertilization. (Healey 1991; Myers et al. 1998; WDNR 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Coho salmon	*Oncorhynchus kisutch*	01–42, 44–48, 50	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes four ESUs of coho salmon in Washington: (1) Lower Columbia River; (2) Southwest Washington; (3) Puget Sound and Strait of Georgia; and (4) Olympic Peninsula. This species is found in a broader diversity of habitats than any of the other native anadromous salmonids. Fry feed primarily on aquatic insects and prefer pools and undercut banks with woody debris; adults feed on herring and other forage fish. **Reproduction/Life History** Coho adults spawn from September to late January, generally in the upper watersheds in gravel free of heavy sedimentation. Developing young remain in gravel for up to 3 months after hatching. Fry emerge from early March to late July. Coho rear in fresh water for 12–18 months before moving downstream to the ocean in the spring. Coho spend between 1 and 2 years in the ocean before returning to spawn. (Groot and Margolis 1991; Murphy and Meehan 1991; WDNR 2005, 2006a; Wydoski and Whitney 2003)
Chum salmon	*Oncorhynchus keta*	01, 03–05, 07–29	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes four ESUs of chum salmon in Washington: (1) Hood Canal summer run; (2) Columbia River; (3) Puget Sound/Strait of Georgia; and (4) Pacific Coast. Little is known about their ocean distribution; maturing individuals that return to Washington streams have primarily been found in the Gulf of Alaska. Chum migrate into rivers and streams of Washington coast, Hood Canal, Strait of Juan de Fuca, Puget Sound, and the Columbia River basin to spawn, but their range does not extend upstream above the Dalles Dam in the Columbia River. Fry feed on chironomid and mayfly larvae, as well as other aquatic insects, whereas juvenile fish in the estuary feed on copepods, tunicates, and euphausiids. **Reproduction/Life History** Chum salmon have three distinct run times: summer, fall and winter. Summer chum begin their upstream migration and spawn from mid-August through mid-October, with fry emergence ranging from the beginning of February through mid-April. Chum fry arrive in estuaries earlier than most salmon, and juvenile chum reside in estuaries longer than most other anadromous species. Chum salmon rear in the ocean for the majority of their adult lives. Fall chum adults enter the rivers from late October through November and spawn in November and December. Winter chum adults migrate upstream from December through January and spawn from January through February. Fall and winter chum fry emerge in March and April and quickly emigrate to the estuary. Chum salmon utilize the low-gradient (from 1–2 percent grade), sometimes tidally influenced lower reaches of streams for spawning. (Healey 1982; Johnson et al. 1997; Quinn 2005; Salo 1991; WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Pink salmon	*Oncorhynchus gorbuscha*	01, 03–05, 07, 09–11, 16–19, 21	1–13	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes two ESUs of pink salmon in Washington, neither of which is listed: (1) Odd-year; and (2) Even-year. The most abundant species of salmon, with 13 stocks identified in Washington. They are the smallest of the Pacific salmon and mature and spawn on a 2-year cycle in Washington (primarily spawning during odd years). Adults are opportunistic feeders in marine habitat, foraging on a variety of forage fish, crustaceans, ichthyoplankton, and zooplankton. Juveniles primarily feed on small crustaceans such as euphausiids, amphipods, and cladocerans. **Reproduction/Life History** Pink salmon will spawn in rivers with substantial amounts of silt. Spawning occurs from August through October. Fry emerge from their redds in late February to early May, depending on water temperature, and migrate downstream to the estuary within 1 month. Juveniles remain in estuarine or nearshore waters for several months before moving offshore as they migrate to the Pacific Ocean, where they remain approximately 1 year until the next spawning cycle. (Hard et al. 1996; Heard 1991; WDNR 2005, 2006a)
Sockeye salmon	*Oncorhynchus nerka*	01, 03–05, 07–11, 16, 19–22, 25–33, 35–37, 40, 41, 44–50	5, 8, 14	**General Information (Habitats and Feeding/Life-history Types)** NOAA Fisheries recognizes seven ESUs of sockeye salmon in Washington: (1) Snake river; (2) Ozette Lake; (3) Baker river; (4) Okanogan River; (5) Quinault Lake; (6) Lake Pleasant; and (7) Lake Wenatchee. WDFW recognizes an additional sockeye salmon stock in the Big Bear Creek drainage of Lake Washington. Kokanee (landlocked sockeye) occur in many lakes, with the larger populations in Banks and Loon lakes in eastern Washington and Lake Whatcom and Lake Washington-Sammamish in western Washington. Juveniles feed on zooplankton, and adults primarily feed on fish, euphausiids, and copepods. **Reproduction/Life History** Spawn in shallow, gravelly habitat in rivers and lakes during August to October. Juvenile sockeye rear in lakes for 1–2 years before migrating to the ocean. Emergence occurs within 3–5 months. (Gustafson et al. 1997; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Steelhead	*Oncorhynchus mykiss*	01, 03–05, 07–12, 14, 15, 17–41, 44–50	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes 15 Distinct Population Segments (DPSs) of steelhead, seven of which occur in Washington. During their ocean phase, steelhead are generally found within 10 and 25 miles of the shore; steelhead remain in the marine environment 2–4 years before returning to fresh water to spawn. Most steelhead spawn at least twice in their lifetimes. Escape cover, such as logs, undercut banks, and deep pools, is important for adult and young steelhead in the freshwater systems. The coastal west-side streams typically support more winter steelhead populations. **Reproduction** A summer spawning run enters fresh water in August and September, and a winter run occurs from December through February. Summer steelhead usually spawn farther upstream than winter populations and dominate inland areas such as the Columbia Basin. Spawning occurs from March to April for both winter and summer run steelhead. After hatching and emergence (approximately 3 months), juveniles establish territories, feeding on microscopic aquatic organisms and then larger organisms such as isopods, amphipods, and aquatic and terrestrial insects. Steelhead rear in fresh water for up to 4 years before migrating to sea. (Busby et al. 1996; McKinnell et al. 1997; WDNR 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Coastal cutthroat trout	*Oncorhynchus clarki clarki*	01–05, 07–30	All	**General Information (Habitats and Feeding/Life-history Types)** NOAA Fisheries has recognized three ESUs in Washington: (1) Puget Sound; (2) Olympic Peninsula; (3) Southwestern Washington/Columbia River. USFWS has assumed sole jurisdiction for this species. No coastal cutthroat trout DPSs are listed under the ESA in Washington. Coastal cutthroat trout exhibit varied life-history forms including: • Resident (stays in streams after rearing in their natal streams) – Resident coastal cutthroat trout utilize small headwater streams for all of their life stages. • Fluvial (migrates to larger rivers after rearing in their natal streams). • Adfluvial (migrates to lakes after rearing in their natal streams). • Anadromous (utilizes estuaries and nearshore habitat but has been caught offshore). Juveniles of all life forms feed primarily on aquatic invertebrates but are opportunistic feeders; adults tend to feed on smaller fish, amphibians, and crustaceans while foraging within the nearshore environment. **Reproduction/Life History** Coastal cutthroat trout are repeat spawners, and juveniles typically rear in the natal streams for up to 2 years. Spawning occurs from late December to February, with incubation lasting approximately 2–4 months. Emergence occurs after 4 months. (Johnson et al. 1999; Pauley et al. 1988; WDNR 2006a)
Redband trout	*Oncorhynchus mykiss gardnerii*	37–40, 45–49, 54–57	NA	**General Information (Habitats and Feeding)** Redband trout is a subspecies of rainbow trout found east of the Cascade Mountains, which prefer cool water that is less than 70°F (21°C), and occupy streams and lakes with high amounts of dissolved oxygen. Their food primarily consists of Daphnia and chironomids as well as fish eggs, fish, and insect larvae and pupae. **Reproduction/Life History** Spawn in streams with clean, small gravel from March through May. Incubation takes approximately 1–3 months, with emergence occurring between June and July. (USFS 2007)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Westslope cutthroat trout	*Oncorhynchus clarki lewisii*	37–39, 44–55, 58–62	NA	**General Information (Habitats and Feeding/Life-history Types)** Cutthroat trout tend to thrive in streams with extensive pool habitat and cover. The westslope is a subspecies of cutthroat trout with three possible life forms: • Adfluvial (migrates to lakes) • Fluvial (migrates to larger rivers) • Resident (stays in streams). The headwater tributaries used by resident cutthroat are typically cold, nutrient-poor waters that result in slow growth. Fluvial and adfluvial forms can exhibit more growth due to warmer water temperatures and nutrient availability. Fry feed on zooplankton, and fingerlings feed on aquatic insect larvae. Adults feed on terrestrial and aquatic insects. **Reproduction/Life History** Spawning: all three life forms spawn in small gravel substrates of tributary streams in the spring (March to July) when water temperature is about 50°F (10°C); incubation occurs during April to August, and emergence occurs from May through August. Fry spend 1–4 years in their natal stream before migrating to their ultimate habitat. (Liknes and Graham 1988; Shepard et al. 1984; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Bull trout	*Salvelinus confluentus*	01, 03–05, 07–23, 26, 27, 29–41, 44–55, 57–62	All	**General Information (Habitats and Feeding/Life-History Types)** Widely distributed in Washington; exhibit four life-history types: • Resident (stays in streams after rearing in their natal streams) • Fluvial (migrates to larger rivers after rearing in their natal streams) • Adfluvial (migrates to lakes after rearing in their natal streams) • Anadromous (bull trout in the nearshore ecosystem rely on estuarine wetlands and favor irregular shorelines with unconsolidated substrates). Young of the year occupy side channels, with juveniles in pools, runs, and riffles; adults occupy deep pools. Juvenile diet includes larval and adult aquatic insects; subadults and adults primarily feed on fish. **Reproduction/Life History** The migratory forms of bull trout, such as anadromous, adfluvial, and fluvial, move upstream by early fall to spawn in September and October (November at higher elevations). Although resident bull trout are already in stream habitats, they move upstream looking for suitable spawning habitat. They prefer clean, cold water (50°F [10°C]) for spawning. Colder water (36–39°F [2–4°C]) is required for incubation. Preferred spawning areas often include groundwater infiltration. Extended incubation periods (up to 220 days) make eggs and fry particularly susceptible to increases in fine sediments. Bull trout typically rear in natal streams for 2–4 years, although resident fish may remain in these streams for their entire lives; multiple life-history forms may occur in the same habitat environments. (Goetz et al. 2004; WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Dolly Varden	*Salvelinus malma*	01, 03, 05, 07, 17–22, 24	6–10, 14–17	**General Information (Habitats and Feeding/Life-History Types)** Species restricted to coastal areas and rivers that empty into them. Juveniles extensively use instream cover; while in the marine systems, they use beaches of sand and gravel. Prefer pool areas and cool temperatures. Feed opportunistically on aquatic insects, crustaceans, salmon eggs, and fish. Closely related to bull trout and exhibit the same life-history traits. Four life-history types occur: • Resident (stays in streams after rearing in their natal streams) • Fluvial (migrates to larger rivers after rearing in their natal streams) • Adfluvial (migrates to lakes after rearing in their natal streams) • Anadromous (migrates to marine waters after rearing in their natal streams). **Reproduction/Life History** Spawn and rear in streams from mid-September through November. Incubation lasts approximately 130 days. Juveniles can spend 2–4 years in their natal streams before migration to marine waters. (Leary and Allendorf 1997; WDNR 2005; Wydoski and Whitney 2003)
Pygmy whitefish	*Prosopium coulteri*	08, 19, 39, 47, 49, 53, 55, 58, 59, 62	NA	**General Information (Habitats and Feeding)** In Washington, pygmy whitefish occur at the extreme southern edge of their natural range; pygmy whitefish were once found in at least 15 Washington lakes but have a current distribution in only nine. They occur most often in deep, oligotrophic lakes with temperatures less than 50°F (10°C), where they feed on zooplankton, such as cladocerans, copepods, and midge larvae. **Reproduction/Life History** Pygmy whitefish spawn in streams or lakes from July through November. They prefer pools, shallow riffles, and pool tail-outs when spawning in streams. Lake spawning by pygmy whitefish occurs at night. Spawning occurs by scattering their eggs over coarse gravel. Incubation and emergence timing are unknown, but eggs are believed to hatch in the spring. (Hallock and Mongillo 1998; WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Olympic mudminnow	*Novumbra hubbsi*	08–24	NA	**General Information (Habitats and Feeding)** Occur in the southern and western lowlands of the Olympic Peninsula, the Chehalis River drainage, lower Deschutes River drainage, south Puget Sound lowlands west of the Nisqually River, and in King County. They are generally found in quiet water with mud substrate, preferring bogs and swamps with dense aquatic vegetation. Mudminnows feed on annelids, insects, and crustaceans. **Reproduction/Life History** Adults spawn from November through June (peaking in April and May). Females deposit eggs onto vegetation where fry remain firmly attached for approximately 1 week after hatching. Incubation lasts approximately 8-10 days. (Harris 1974; Mongillo and Hallock 1999; WDNR 2005, 2006a)
Lake chub	*Couesius plumbeus*	48, 61; other locations unknown	NA	**General Information (Habitats and Feeding)** Bottom dwellers inhabiting a variety of habitats in lakes and streams, but are known to prefer small, slow streams. In Washington, they are known only from the northeastern part of the state (small streams and lakes in Okanogan and Stevens counties). Juveniles feed on zooplankton and phytoplankton, whereas adults primarily feed on insects. **Reproduction/Life History** Lake chub move into shallow areas on rocky and gravelly substrates in tributary streams of lakes or lakeshores during the spring to spawn when water temperatures are between 55 and 65°F (13 and 18°C). The eggs are broadcast over large rocks and then settle into the smaller substrate, hatching after approximately 10 days. (WDNR 2005; Wydoski and Whitney 2003)
Leopard dace	*Rhinichthys falcatus*	25–31, 37–41, 44–50	NA	**General Information (Habitats and Feeding)** In Washington, leopard dace inhabit the bottoms of streams and small to mid-sized rivers, specifically the Columbia, Snake, Yakima, and Simikameen Rivers, with velocities less than 1.6 ft/sec (0.5 m/sec); prefer gravel and small cobble substrate covered by fine sediment with summer water temperatures ranging between 59 and 64°F (15 and 18°C). Juveniles feed primarily on aquatic insects; adult leopard dace consume terrestrial insects. **Reproduction/Life History** Breeding habitat for dace generally consists of the gravel or cobble bottoms of shallow riffles; leopard dace breed in slower, deeper waters than the other dace species. The spawning period for dace is from May through July. The eggs adhere to rocky substrates. Fry hatch approximately 6–10 days after fertilization, and juveniles spend 1–3 months rearing in shallow, slow water. (WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Margined sculpin	*Cottus marginatus*	32, 35	NA	**General Information (Habitats and Feeding)** Endemic to southeastern Washington (smaller tributary streams of the Walla Walla and Tucannon River drainages) where habitat is in deeper pools and slow-moving glides in headwater tributaries with silt and small gravel substrate. They prefer cool water less than 68°F (20°C) and avoid high-velocity areas. Food includes immature aquatic insects, invertebrates, small fish, and eggs. **Reproduction/Life History** Spawning occurs in May and June primarily under rocks, root wads, or logs. The female deposits a mass of adhesive eggs in the nest, which is guarded by the male. Incubation duration unknown. (Mongillo and Hallock 1998; WDNR 2005; Wydoski and Whitney 2003)
Mountain sucker	*Catostomus platyrynchus*	25–35, 37–41, 44–50	NA	**General Information (Habitats and Feeding)** Distribution restricted to Columbia River system. Found in clear, cold mountain streams less than 40 ft wide and in some lakes; prefer deep pools in summer with moderate current. Food consists of algae and diatoms. Juveniles prefer slower side channels or weedy backwaters. **Reproduction/Life History** Males reach sexual maturity in 2–3 years and females in 4 years. Spawning in June and July when water temperatures exceed 50°F (10°C). Spawning occurs in gravelly riffles of small streams when suckers move into those reaches to feed on algae. Spawning likely occurs at night when water temperatures are in a range of 51–66°F (10.5–19°C). Fertilized eggs fall into and adhere to the spaces between the gravel composite. Incubation period lasts approximatley 8-14 days. (Wydoski and Whitney 2003)
Umatilla dace	*Rhinichthys umatilla*	31, 36–41, 44–50, 59–61	NA	**General Information (Habitats and Feeding)** Umatilla dace are benthic fish found in relatively productive, low-elevation streams with clean substrates of rock, boulders, and cobbles in reaches where water velocity is less than 1.5 ft/sec (0.5 m/sec). Feeding is similar to that described for leopard dace. Juveniles occupy streams with cobble and rubble substrates, whereas adults occupy deeper water habitats. **Reproduction/Life History** Spawning behaviors are similar to those described for leopard dace, with spawning primarily occurring from early to mid-July. (WDNR 2005, 2006a; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Pacific lamprey	*Lampetra tridentata*	01, 03–05, 07–35, 37–40, 44–50	All	**General Information (Habitats and Feeding)** Found in most large coastal and Puget Sound rivers and Columbia, Snake, and Yakima river basins. The larvae are filter feeders, residing in mud substrates and feeding on algae and other organic matter for at least 5 years. **Reproduction/Life History** From July through October, maturing Pacific lamprey enter fresh water and gradually move upstream to spawn the following spring. The nest usually consists of a shallow depression built in gravel and rock substrates. Eggs hatch in 2–4 weeks, with newly hatched larvae remaining in the nest for 2–3 weeks before moving downstream as larvae (ammocoetes). Juveniles migrate to the Pacific Ocean 4–7 years after hatching and attach to fish in the ocean for 20–40 months before returning to rivers to spawn. (WDNR 2005; Wydoski and Whitney 2003)
River lamprey	*Lampetra ayresi*	01, 03, 05, 07–16, 20–40	1–9, 11–17	**General Information (Habitats and Feeding)** Detailed distribution records are not available for Washington, but they are known to inhabit coastal rivers, estuaries, and the Columbia River system. They have also been observed in Lake Washington and its tributaries. In the marine system, river lamprey inhabit nearshore areas. Adults are anadromous living in the marine system as parasites on fish. Adult river lamprey are believed to occupy deep portions of large river systems. The larvae feed on microscopic plants and animals. **Reproduction/Life History** Adults migrate back into fresh water in the fall. Spawning occurs in winter and spring. Eggs hatch in 2–3 weeks after spawning. Juveniles are believed to migrate from their natal rivers to the Pacific Ocean several years after hatching; adults spend 10–16 weeks between May and September in the ocean before migrating to fresh water. (WDNR 2005; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Western brook lamprey	*Lampetra richardsoni*	01, 03, 05, 07–14, 16, 20–40	NA	**General Information (Habitats and Feeding)** Found in small coastal and Puget Sound rivers and lower Columbia and Yakima river basins; spends entire life in fresh water. Adults are found in cool water (52–64°F [11–17.8°C]) on pebble/rocky substrate. Larvae (ammocoetes) are filter feeders, consuming primarily diatoms. Adults do not feed and die within a month of spawning. **Reproduction/Life History** Spawning generally occurs from April through July, with adults creating nests in coarse gravel at the head of riffles. Eggs hatch after about 10 days in water between 50 and 60°F (10 and 16°C). Within 30 days of hatching, ammocoetes emerge from the nests and move to the stream margin, where they burrow into silty substrates. Larvae remain in the stream bottom—apparently moving little—for approximately 4–6 years. (Wydoski and Whitney 2003)
Green sturgeon	*Acipenser medirostris*	22, 24, 28	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes two DPSs of green sturgeon, both of which can be found in Washington. The southern DPS is listed as threatened and the northern DPS is a species of concern. Habits and life history not well known. Washington waters with green sturgeon populations include the Columbia River, Willapa Bay, and Grays Harbor, in addition to marine waters. They spend much of their life in marine nearshore waters and estuaries feeding on fishes and invertebrates. **Reproduction/Life History** Spawning generally occurs in spring in deep, fast-flowing sections of rivers. Spawning habitat includes cobble or boulder substrates. Green sturgeon move upstream during spring to spawn and downstream during fall and winter. Large eggs sink to bottom. (Adams et al. 2002; Emmett et al. 1991; Kynard et al. 2005; Nakamoto and Kisanuki 1995; Wydoski and Whitney 2003)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
White sturgeon	*Acipenser transmontanus*	01, 03, 05–22, 24–37, 40–42, 44–61	All	**General Information (Habitats and Feeding)** Found in marine waters and major rivers in Washington, including the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. In marine environments, adults and subadults use estuarine and marine nearshore habitats, including some movement into intertidal flats to feed at high tide. Some landlocked populations exist behind dams on the Columbia River. Juveniles feed on mysid shrimp and amphipods; large fish feed on variety of crustaceans, annelid worms, mollusks, and fish. **Reproduction/Life History** Spawn in deep, fast-flowing sections of rivers (prefer swift [2.6–9.2 ft/sec (0.8–2.8 m/sec)] and deep [13–66 ft (4–20 m)] water) on bedrock, cobble, or boulder substrates. Spawning occurs from April through July, with incubation lasting approximately 7 days and emergence following in another 7 days. (Emmett et al. 1991; WDNR 2005; Wydoski and Whitney 2003)
Eulachon	*Thaleichthys pacificus*	01–29 (mouths of major rivers)	14–17	**General Information (Habitats and Feeding)** Eulachon occur from northern California to southwestern Alaska in offshore marine waters. They are plankton-feeders, eating crustaceans such as copepods and euphausiids; larvae and post larvae eat phytoplankton and copepods. They are an important prey species for fish, marine mammals, and birds. **Reproduction/Life History** Spawn in tidal portions of rivers in spring when water temperature is 40–50°F (4–10°C), generally from March through May; use a variety of substrates, but sand and gravel are most common. Eggs stick to substrate and incubation ranges from 20–40 days (dependent on temperature). Larvae drift downstream to salt water where juveniles rear in nearshore marine areas. (Howell et al. 2001; Langer et al. 1977; Lewis et al. 2002; WDFW 2001; WDNR 2005; Willson et al. 2006)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Longfin smelt	*Spirinchus thaleichthys*	01–03, 05–17, 22 and 24	1–9, 15–17	**General Information (Habitats and Feeding)** Marine species that spawns in streams not far from marine waters. They are anadromous, with some populations in Lake Washington that spawn in tributaries, including the Cedar River. Juveniles use nearshore habitats and a variety of substrates; juveniles feed on zooplankton. Adults feed on copepods and euphausiids. Most adults die after spawning. **Reproduction** Spawn in coastal rivers from October through December. Lake Washington populations spawn from January through April. Eggs hatch in approximately 40 days and the larvae drift downstream to salt water. (Gotthardt 2006; WDNR 2005; Wydoski and Whitney 2003)
Pacific sand lance	*Ammodytes hexapterus*	NA	All	**General Information (Habitats and Feeding)** Widespread in Puget Sound, Strait of Juan de Fuca, and coastal estuaries. Schooling plankton feeders. Adults feed during the day and burrow into the sand at night. **Reproduction/Life History** Spawn on sand and beaches with gravel up to 1-inch in diameter at tidal elevations of +4–5 ft (+1.5 meters) to approximately the mean higher high water (MHHW) line from November through February. Emergence occurs from January to April. Larvae and young rear in bays and nearshore areas. (Garrison and Miller 1982; Nightingale and Simenstad 2001b; NRC 2001; Penttila 2000; Penttila 2001; WDFW 1997a)
Surf smelt	*Hypomesus pretiosus*	NA	All	**General Information (Habitats and Feeding)** Schooling plankton-feeding forage fish. They feed on a variety of zooplankton, planktonic crustaceans, and fish larvae. Adult surf smelt are pelagic but remain in nearshore habitats. Juveniles rear in nearshore areas, and adults form schools offshore; feed on planktonic organisms. Also an important forage fish. **Reproduction/Life History** Spawning occurs year-round in north Puget Sound, fall and winter in south Puget Sound, and summer along the coast. They spawn at the highest tides during high slack tide on coarse sand and pea gravel. Incubation is 2–5 weeks. Emergence varies with season: 27–56 days in winter, 11–16 days in summer. (Nightingale and Simenstad 2001b; NRC 2001; Penttila 2000; Penttila 2001; WDFW 1997c)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Pacific herring	*Clupea harengus pallasi*	NA	1, 2, 4, 5, 8–13, 16, 17	**General Information (Habitats and Feeding)** Eighteen separate stocks in Puget Sound. Widely distributed throughout Puget Sound and coastal wetlands and estuaries. Pacific herring adults feed on small fish, copepods, decapod crab larvae, and euphausiids. Juveniles feed primarily on euphausiids, copepods, and small crustacean larvae. Are also an important forage fish. **Reproduction/Life History** Utilize intertidal and subtidal habitats (between 0 and -40 ft [0 and -12.2 m] mean lower low water [MLLW]) for spawning and juvenile rearing; spawning also occurs above MLLW. Spawning occurs from late January to early April. Eggs are adhered to eelgrass, kelp, seaweed, and sometimes on pilings. Eggs hatch after approximately 10 days. Larvae are pelagic. (Nightingale and Simenstad 2001b; Penttila 2000; Simenstad et al. 1979; WDFW 1997b)
Lingcod	*Ophiodon elongatus*	NA	All	**General Information (Habitats and Feeding)** The lingcod is a large top-level carnivore fish found throughout the West Coast of North America. Adult lingcod have a relatively small home range. Juveniles prefer sand habitats near the mouths of bays and estuaries, while adults prefer rocky substrates. Larvae and juveniles are generally found in upper 115 ft (35 m) of water. Adults prefer slopes of submerged banks with macrophytes and channels with swift currents. Larvae feed on copepods and amphipods; juveniles feed on small fishes; and adults on fish, squid, and octopi. **Reproduction/Life History** Spawn in shallow water and intertidal zone from January through late March. Egg masses adhere to rocks, and incubation is from February to June. Larvae spend 2 months in pelagic nearshore habitat. (Adams and Hardwick 1992; Emmett et al. 1991; Giorgi 1981; NMFS 1990; NRC 2001)
Pacific cod	*Gadus macrocephalus*	NA	All	**General Information (Habitats and Feeding)** Pacific cod are widely distributed in relatively shallow marine waters throughout the northern Pacific Ocean (Washington's inland marine waters are considered the southern limit of populations). Adults and large juveniles are found over clay, mud, and coarse gravel bottoms; juveniles use shallow vegetated habitats such as sand-eelgrass. Feed opportunistically on invertebrates (worms, crabs, shrimp) and fishes (sand lance, pollock, flatfishes). Larvae feed on copepods, amphipods, and mysids. **Reproduction/Life History** Broadcast spawners during late fall through early spring. Eggs sink and adhere to the substrate. Incubate for 1–4 weeks, and larvae spend several months in the water column. Juvenile cod metamorphose and settle to shallow vegetated habitats. (Albers and Anderson 1985; Bargmann 1980; Dunn and Matarese 1987; Garrison and Miller 1982; Hart 1973; Nightingale and Simenstad 2001b; NMFS 1990; NRC 2001)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Pacific hake	*Merluccius productus*	NA	All	**General Information (Habitats and Feeding)** Pacific hake are schooling fish. The coastal stock of hake is migratory; Puget Sound stocks reside in estuaries and rarely migrate. Larvae feed on calanoid copepods; juveniles and small adults feed on euphasiids; adults eat amphipods, squid, herring, and smelt. **Reproduction/Life History** Puget Sound spawning occurs from March through May at mid-water depths of 50–350 ft (15–90 m); may spawn more than once per season. Eggs and larvae are pelagic. (Bailey 1982; McFarlane and Beamish 1986; NMFS 1990; NRC 2001; Quirollo 1992)
Walleye pollock	*Theragra chalcogramma*	NA	All	**General Information (Habitats and Feeding)** Widespread species in northern Pacific. Washington is the southern end of their habitat. Larvae and small juveniles are found at 200-ft (60-m) depth; juveniles use nearshore habitats of a variety of substrates. Juveniles feed on small crustaceans, adults feed on copepods, euphausiids, and young pollock. **Reproduction/Life History** Broadcast spawning occurs from February through April. Eggs are suspended at depths ranging from 330–1,320 ft (100–400 m). Pelagic larvae settle near the bottom and migrate to inshore, shallow habitats for their first year. (Bailey et al. 1999; Garrison and Miller 1982; Livingston 1991; Miller et al. 1976; NRC 2001)
Black rockfish	*Sebastes melanops*	NA	All	**General Information (Habitats and Feeding)** Adults prefer deep and shallow rock substrates in summer, deeper water in winter. Kelp and eelgrass are preferred habitat for juveniles that feed on nekton and zooplankton. Adults feed on amphipods, crabs, copepods, and small fish. **Reproduction/Life History** Spawning occurs from February through April; ovoviviparous incubation as with other rockfish species. Larvae are planktonic for 3–6 months, where they are dispersed by currents, advection, and upwelling. They begin to reappear as young-of-the-year fish in shallow, nearshore waters. (Kramer and O'Connell 1995; WDNR 2006a)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Bocaccio rockfish	*Sebastes paucispinis*	NA	All	**General Information (Habitats and Feeding)** Adults semidemersal in shallow water over rocks with algae, eelgrass, and floating kelp. Larvae feed on diatoms; juveniles feed on copepods and euphausiids. **Reproduction/Life History** Ovoviviparous spawning occurs year-round, with incubation lasting 40–50 days. Larvae and juveniles are pelagic. (Garrison and Miller 1982; Hart 1973; Kramer and O'Connell 1995; MBC Applied Environmental Sciences 1987; NRC 2001; Sumida and Moser 1984)
Brown rockfish	*Sebastes auriculatus*	NA	All	**General Information (Habitats and Feeding)** Utilize shallow-water bays with natural and artificial reefs and rock piles; estuaries used as nurseries; can tolerate water temperatures to at least 71°F (22°C); eat small fishes, crabs, and isopods. **Reproduction/Life History** Spawning occurs from March through June. Larvae are released from the female into the pelagic environment in May and June (ovoviviparous incubation). Larvae live in the upper zooplankton layer for up to 1 month before they metamorphose into pelagic juveniles. The pelagic juveniles spend 3–6 months in the water column as plankton. They then settle in shallow water nearshore, later migrating to deeper water. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; Stein and Hassler 1989)
Canary rockfish	*Sebastes pinniger*	NA	All	**General Information (Habitats and Feeding)** Adults use sharp drop-offs and pinnacles with hard bottoms; often associated with kelp beds; feed on krill and occasionally on fish. Adults are mostly found at depths of 260–660 ft (80–200 meters) (with two recorded at 2,750 ft [838 meters]), tending to collect in groups around pinnacles and similar high-relief rock formations, especially where the current is strong. Young canary rockfish live in relatively shallow water, moving to deeper water as they mature. Juveniles feed on small crustacea such as krill larvae (and eggs), copepods, and amphipods, while adults eat krill and small fish. **Reproduction/Life History** Spawning is ovoviviparous and occurs from January through March. Larvae and juveniles are pelagic. (Boehlert 1980; Boehlert and Kappenman 1980; Boehlert et al. 1989; Hart 1973; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; Sampson 1996)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
China rockfish	*Sebastes nebulosis*	NA	All	**General Information (Habitats and Feeding)** Occur inshore and on open coast in sheltered crevices. Feed on crustacea (brittle stars and crabs), octopi, and fish. Juveniles are pelagic, but the adults are sedentary associating with rocky reefs or cobble substrates. **Reproduction/Life History** Spawning occurs from January through July; ovoviviparous incubation as with other rockfish species. Individual China rockfish spawn once a year. Larvae settle out of the plankton between 1 and 2 months after release. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; Rosenthal et al. 1988)
Copper rockfish	*Sebastes caurinus*	NA	All	**General Information (Habitats and Feeding)** Occur both inshore and on open coast; adults prefer rocky areas in shallower water than other rockfish species. Juveniles use shallow and nearshore macrophytes and eelgrass habitat; feed on crustaceans, fish, and mollusks. **Reproduction/Life History** Spawning occurs from March through May, with ovoviviparous incubation from April to June. Larvae are pelagic in deeper water before moving inshore. Newly spawned fish begin settling near the surface around large algae canopies or eelgrass, when available, or closer to the bottom when lacking canopies. (Eschmeyer et al. 1983; Haldorson and Richards 1986; Kramer and O'Connell 1995; Matthews 1990; NRC 2001; Stein and Hassler 1989)
Greenstriped rockfish	*Sebastes elongates*	NA	All	**General Information (Habitats and Feeding)** Adults found in benthic and mid-water columns. They live at between 330 and 825 ft (100 and 250 m). As they age, greenstriped rockfish move to deeper water. They are solitary and are often found resting on the seafloor and living among cobble, rubble, or mud. Adults feed on euphausiids, small fish, and squid. **Reproduction/Life History** From 10,000 to over 200,000 eggs are produced by the females each season by ovoviviparous spawning. Greenstriped rockfish release one brood of larvae in Washington. Larval release varies, occurring generally from January through July, depending on geographic location. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Quillback rockfish	*Sebastes maliger*	NA	All	**General Information (Habitats and Feeding)** Shallow-water benthic species in inlets near shallow rock piles and reefs. Juveniles use eelgrass, sand, and kelp beds. Feed on amphipods, crabs, and copepods. **Reproduction/Life History** Ovoviviparous spawning from April through July, with larval release from May to July. (Kramer and O'Connell 1995; WDNR 2006a)
Redstripe rockfish	*Sebastes proriger*	NA	All	**General Information (Habitats and Feeding)** Adults found from 330- to 1,000-ft (100- to 300-m) depths, and young often found in estuaries in high- and low-relief rocky areas. Juveniles feed on copepods and euphausiids; adults eat anchovies, herring, and squid. **Reproduction/Life History** Spawning is ovoviviparous, occurring from January through March. Larvae and juveniles are pelagic. (Garrison and Miller 1982; Hart 1973; Kendall and Lenarz 1986; Kramer and O'Connell 1995; NRC 2001; Starr 1996)
Tiger rockfish	*Sebastes nigrocinctus*	NA	All	**General Information (Habitats and Feeding)** Semidemersal to demersal species occurring at depths ranging from shallows to 1,000 ft (305 m); larvae and juveniles occur near surface and range of depth; adults use rocky reefs, canyons, and headlands; generalized feeders on shrimp, crabs, and small fishes. **Reproduction/Life History** Ovoviviparous spawning peaks in May and June. Juveniles are pelagic. (Garrison and Miller 1982; Kramer and O'Connell 1995; Moulton 1977; NRC 2001; Rosenthal et al. 1988)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Widow rockfish	*Sebastes entomelas*	NA	All	**General Information (Habitats and Feeding)** Adults found from 330- to 1,000-ft (100- to 300-m) depths near rocky banks, ridges, and seamounts; adults feed on pelagic crustaceans, Pacific hake, and squid; juveniles feed on copepods and euphausiids. **Reproduction /Life History** Ovoviviparous spawning occurs from October through December. One brood of 95,000 to 1,113,000 eggs are produced by female widows per year. The season of larval release occurs earlier in the southern parts of their range than in the northern regions, likely January through April in Washington waters. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Laroche and Richardson 1981; NMFS 1990; NRC 2001; Reilly et al. 1992)
Yelloweye rockfish	*Sebastes ruberrimus*	NA	All	**General Information (Habitats and Feeding)** Adults are found from depths of 80–1,800 ft (24–550 m), near reefs and cobble bottom. Juveniles prefer shallow, broken-bottom habitat. Juveniles often hide in rock crevices; adults are demersal and solitary, tending to remain localized and not making extensive migrations. Adults feed on other rockfish species, sand lance, herring, shrimp, rock crabs, and snails. **Reproduction/Life History** Ovoviviparous spawning in late fall or early winter, with the larvae released from May to July. (Eschmeyer et al. 1983; Hart 1973; Kramer and O'Connell 1995; NRC 2001; Rosenthal et al. 1988)
Yellowtail rockfish	*Sebastes flavidus*	NA	All	**General Information (Habitats and Feeding)** Adults found from 165- to 1,000-ft (50- to 300-m) depths; adults semipelagic or pelagic over steep-sloping shores and rocky reefs. Juveniles occur in nearshore areas. Adults are opportunistic feeders on pelagic animals including hake, herring, smelt, squid, krill, and euphausiids. **Reproduction/Life History** Ovoviviparous spawning from October through December. Incubation is between January and March. Larvae and juveniles are pelagic swimmers. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; O'Connell and Carlile 1993)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Olympia oyster	*Ostrea lurida*	NA	1–14, 17	**General Information (Habitats and Feeding)** Species found throughout the inland waters of Puget Sound, as well as in Willapa Bay and possibly Grays Harbor; also grown commercially in Puget Sound. They occupy nearshore ecosystem on mixed substrates with solid attachment surfaces and are found from 1 ft (0.3 m) above MLLW to 2 ft (0.6m) below MLLW. Intolerant of siltation. **Reproduction/Life History** Reproduce spring to fall when water temperatures are between 54 and 61°F (12.5 and 16°C) by broadcast spawning. After 8–12 days, larvae develop into free-swimming larvae. Larvae are free-swimming for 2–3 weeks before they settle onto hard substrate, such as oyster shells and rocks. (Baker 1995; Couch and Hassler 1990; West 1997)
Northern abalone	*Haliotis kamtschatkana*	NA	10	**General Information (Habitats and Feeding)** Also known as pinto abalone. Presence in Washington is limited to the Strait of Juan de Fuca and the San Juan Islands. Occupies bedrock and boulders from extreme low water to 100 ft (30 m) below MLLW; usually associated with kelp beds. The abalone is completely vegetarian and uses its radula to scrape pieces of algae from the surface of rocks. **Reproduction/Life History** Broadcast spawners that release pelagic gametes that develop into free-swimming larvae using cilia to propel themselves. After up to a week, the larvae settle to the bottom, shed their cilia, and start growing a shell to begin sedentary adult life on crustose coralline algae. (Gardner 1981; NMFS 2007a; WDNR 2006b; West 1997)
Newcomb's littorine snail	*Algamorda subrotundata*	NA	14–17	**General Information (Habitats and Feeding)** Found in Grays Harbor and Willapa Bay on Washington coast; current distribution uncertain. Algae feeder occupying narrow band in *Salicornia* salt marshes above MHHW and is not considered a true marine gastropod. **Reproduction/Life History** Broadcast spawning in salt marshes. Other reproductive information unknown. (Larsen et al. 1995)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Giant Columbia River limpet	*Fisherola nuttalli*	35, 36, 40, 45, 47–49	NA	**General Information (Habitats and Feeding)** Also known as the shortface lanx, it occupies fast-moving and well-oxygenated streams. It is found in the Hanford Reach segment of the Columbia River, Wenatchee, Deschutes (OR), Okanogan, Snake, and Methow rivers. Prefers shallow, rocky areas of cobble to boulder substrates and diatom-covered rocks, and feeds by grazing on algae attached to rocks. **Reproduction/Life History** Broadcast external fertilization. Reproduction timing is unknown. (Neitzel and Frest 1989; Neitzel and Frest 1990; Pacific Biodiversity Institute 2007)
Great Columbia River spire snail	*Fluminicola columbiana*	35, 45, 48, 49; other locations unknown	NA	**General Information (Habitats and Feeding)** Also known as the Columbia pebblesnail and ashy pebblesnail, its current range is restricted to rivers, streams, and creeks of the Columbia River basin. It requires clear, cold streams with highly oxygenated water and is generally found in shallow water (less than 5 inches [13 cm] deep) with permanent flow on cobble-boulder substrates. Spire snails live on and under rocks and vegetation in the slow to rapid currents of streams where they graze on algae and small crustaceans. **Reproduction/Life History** They are short-lived, usually reaching sexual maturity within a year, at which time they breed and die**.** Unknown reproduction timing. (Neitzel and Frest 1989; Neitzel and Frest 1990; Pacific Biodiversity Institute 2007)
California floater (mussel)	*Anodonta californiensis*	30, 36, 37, 40, 42, 47–49, 52–54, 58–61	NA	**General Information (Habitats and Feeding)** In Washington, it is known to occur in the Columbia and Okanogan rivers and several lakes. Freshwater filter feeder requiring clean, well-oxygenated water for survival that is declining throughout much of its historical range. California floater mussels are intolerant of habitats with shifting substrates, excessive water flow fluctuations, or seasonal hypoxia. **Reproduction/Life History** Spring spawning occurs after adults reach 6–12 years in age. Fertilization takes place within the brood chambers of the female mussel. Fertilized eggs develop into a parasitic stage called glochidia, which attach to species-specific host fish during metamorphosis. After reaching adequate size, juvenile mussels release from the host and attach to gravel and rocks. (Box et al. 2003; Frest and Johannes 1995; Larsen et al. 1995; Nedeau et al. 2005; Watters 1999; WDNR 2006b)

Table 5-1 (continued). Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Western ridged mussel	*Gonidea angulata*	01, 03–05, 07–11, 13, 21–42, 44–55, 57–62	NA	**General Information (Habitats and Feeding)** Specific information on this species is generally lacking; reside on substrates ranging from firm mud with the presence of some sand, silt, or clay to coarse gravel in creeks, streams, and rivers. They require constant, well-oxygenated flow, and shallow water (<10 ft [3 m] depth). This species may tolerate seasonal turbidity but is absent from areas with continuous turbidity and is sensitive to water quality changes such as eutrophication or presence of heavy metals. **Reproduction/Life History** During breeding, males release sperm into the water and females must bring this into their shell for fertilization to occur. Larvae called glochidia are released by the female and attach to the gills of fish for 1–6 weeks; postlarval mussels hatch from cysts as free-living juveniles to settle and bury in the substrate. (COSEWIC 2003; WDNR 2006b)

Source: Modified from (Jones & Stokes 2006).

[a] Water Resource Inventory Areas (WRIAs) are administration and planning boundaries for watershed areas, as established and managed by the Washington State Department of Ecology (Ecology). WRIA designations were formalized under WAC 173-500-040 and authorized under the Water Resources Act of 1971, Revised Code of Washington (RCW) 90.54. For WRIA boundary locations and related information, see URL = http://www.ecy.wa.gov/services/gis/maps/wria/wria.htm.

[b] Tidal Reference Areas as follows (from WAC 220-110-240): 1 = Shelton, 2 = Olympia, 3 = South Puget Sound, 4 = Tacoma, 5 = Seattle, 6 = Edmonds, 7 = Everett, 8 = Yokeko Point, 9 = Blaine, 10 = Port Townsend, 11 = Union, 12 = Seabeck, 13 = Bangor, 14 = Ocean Beaches, 15 = Westport, 16 = Aberdeen, 17 = Willapa Bay.

6.0 Conceptual Framework for Assessing Impacts

Flow control structures are located throughout Washington State. The placement of these structures and the uses associated with them will affect, to varying degrees, the controlling factors of the aquatic ecosystem in which they are located. In this white paper, an **impact** is defined as an unnatural disturbance to habitat-controlling factors such as light, wave energy, substrate, water quality parameters, littoral drift, or channel geomorphology. These controlling factors determine various aspects of the habitat structure (e.g., sand or cobble substrates or eelgrass or kelp). For example, the habitat structure provided by shoreline overhanging vegetation can provide shade for species using nearshore shallow water and upper beach habitats. This shade serves the ecological function of regulating temperature and supporting the foodweb through organic litter and insect input. Figure 6-1 illustrates the conceptual framework used in this white paper to identify impacts on HCP species and their habitats from flow control structures.

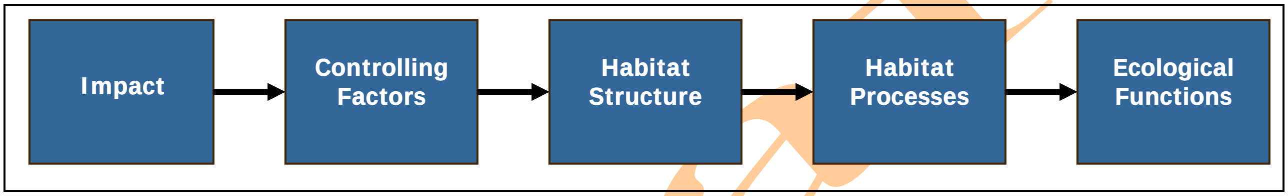


Figure 6-1. Conceptual framework for assessing impacts (Williams and Thom 2001).

Table 6-1 identifies the **mechanisms of impact** that are known to be associated with flow control structures covered in this white paper. This white paper presents what is known about the effects of these mechanisms on HCP species. By identifying these impacts and the nature of the risks these impacts exert on HCP species, measures can be implemented to avoid and, if avoidance is not possible, to minimize harmful impacts on these species and the habitats that support their growth and survival.

The identification of impact mechanisms associated with HPA-authorized activities that affect habitat is based on a model described by Williams and Thom (2001). For analyzing risk of take and refining the impact analysis as it pertains directly to listed species or species that will be addressed in the HCP, the “exposure-response” model developed by the USFWS was used (National Conservation Training Center 2004). Each of these models is discussed in more detail below.

The Williams and Thom model provides the framework for analysis based on the literature search (as described in Section 3 [*Methods*]). The goals of this framework are to:

- Elucidate impacts associated with each HPA subactivity
- Determine how those impacts manifest in effects on habitat and habitat functions utilized by the HCP species.

- Develop recommendations for impact avoidance, minimization, and mitigation measures that target the identified impacts.

Table 6-1. Impact mechanisms and submechanisms associated with flow control structures.

Impact Mechanism	Submechanisms
Construction and Maintenance Activities	Elevated Underwater Noise Bank/Channel/Shoreline Disturbance Contamination From Chemical and Fuel Spills Dewatering, Flow Bypass, and Fish Handling Channel Rewatering Construction and Maintenance Dredging
Hydraulic and Geomorphic Modifications	Altered Flow Regime Altered Channel Geometry Altered Sediment Transport Altered Substrate Composition Altered Groundwater–Surface Water Interactions
Ecosystem Fragmentation	Altered Longitudinal Connectivity Altered River-Floodplain Connectivity Altered Groundwater–Surface Water Interactions Altered Large Woody Debris (LWD) Transport and Recruitment Altered Community Composition
Riparian Vegetation Modifications	Altered Shading, Solar Input, and Ambient Air Temperature Altered Bank and Shoreline Stability Altered Allochthonous Inputs Altered Groundwater–Surface Water Interactions Altered Habitat Complexity Altered Nutrient/Pollutant Loading
Aquatic Vegetation Modifications	Altered Autochthonous Production Altered Habitat Complexity Altered Nutrient/Pollutant Loading
Water Quality Modifications[a]	Altered Temperature Regime Altered Dissolved Oxygen Altered Suspended Solids and Turbidity Increases in Contaminated Sediment Altered pH Levels Altered Nutrient Loading Introduction of Toxic Substances Altered Salinity Metal Toxicity

[a] This list contains all possible water quality modifications; however, only relevant submechanisms for each subactivity are presented in Sections 7.1 through 7.6. In other words, not all of these submechanisms are discussed for each subactivity.

The analysis process begins with an impact that, in this case, would consist of activities authorized under an HPA for a flow control project. The impact will exert varying degrees of effect on controlling factors within the ecosystem (Williams and Thom 2001). Controlling factors are those physical processes or environmental conditions (e.g., flow conditions or wave energy) that control local habitat structure (e.g., substrate or vegetation). Habitat structure is linked to habitat processes (e.g., shading or cover), which are linked to ecological functions (e.g., refuge and prey production). These linkages form the "**impact pathway**" in which alterations to the environment associated with HPA-authorized activities can lead to impacts on the ecological function of the habitat for HCP species. **Impact mechanisms** are the alterations to any of the conceptual framework components along the impact pathway that can result in an impact on ecological function(s) and therefore on HCP species.

For each HPA-authorized activity addressed in this white paper, several principal impact mechanisms were identified for each subactivity type, from a geomorphological, engineering, hydrologic, and biological perspective.

This impact analysis serves to identify the direct and indirect impacts that could potentially affect HCP species. To further refine the analysis in each white paper, the exposure-response model (National Conservation Training Center 2004) was incorporated into the impact analysis. The exposure-response model evaluates the likelihood that adverse effects may occur as a result of species exposure to one or more stressors. This model takes into account the life-history stage most likely to be exposed and thereby affected.

The exposure-response model was incorporated as a series of matrices, presented in Appendix A, with results synthesized in Section 7 (*Direct and Indirect Impacts*) and Section 9 (*Potential Risk of Take*) of this white paper. In these species-specific exposure-response matrices, each mechanism and submechanism was initially examined and evaluated to:

- Identify and characterize specific impacts or stressors
- Evaluate the potential for exposure (potential for species to be exposed = identification of stressor timing/duration/frequency/life-history, form, and presence coincident with an impact)
- Identify the species' anticipated response to a stressor
- Identify measures that could reduce exposure
- Identify performance standards if appropriate
- Characterize the resulting effects of specific impacts on the various species.

With regard to exposure, standard language was used to indicate when an impact occurs, and for how long and how frequently the stressor or impact occurs. Definitions of the terms used in this analysis are listed in Table 6-2.

Table 6-2. Definitions of terms used in the exposure-response analysis for this white paper.

Parameter	Description	Exposure	Definition
When	The timing during which stressor exposure occurs (e.g., time of day, season, associated with operations or maintenance)	–	Defined flexibly as appropriate for each stressor.
Duration	The length of time the receptor is expected to be exposed to the stressor	Permanent	Stressor is permanent (e.g., conversion of habitat to built environment)
		Long-term	Stressor will last for greater than five years to decades (e.g., time required for complete riparian recovery)
		Intermediate-term	Stressor will last from 6 months to approximately 5 years (e.g., time required for beach substrate to recover from construction equipment)
		Short-term	Stressor will last from days to 6 months (e.g., time required for invertebrate community to recolonize following dewatering)
		Temporary	Stressor associated with transient action (e.g., pile driving noise)
Frequency	The regularity with which stressor exposure is expected to occur and/or the time interval between exposure	Continuous	Stressor is ongoing and occurs constantly (e.g., permanent modification of habitat suitability)
		Intermittent	Stressor occurs routinely on a daily basis
		Daily	Stressor occurs once per day for extended periods (e.g., daytime structural shading)
		Common	Stressor occurs routinely (i.e., at least once per week or several times per month)
		Seasonal	Stressor occurs for extended periods during specific seasons (e.g., temperature effects occurring predominantly in winter and summer)
		Annual	Stressor occurs annually for a short period of time
		Interannual–decadal	Stressor occurs infrequently (e.g., pile driving associated with project construction and maintenance)

Based on life-history information, an analysis of potential exposure was completed for each species. This included an analysis of the direct and indirect impacts (associated with each of the impact mechanisms) on the different life-history stages of each species and the likely responses of each species to these stressors. Impact minimization measures to reduce or avoid submechanism impacts were also identified. A final conclusion regarding the overall effect of the submechanism/stressor on a species is also presented in Appendix A. Where information was available, the cumulative effects associated with the major impact mechanisms were also identified (see Section 8 [*Cumulative Effects*]).

The information generated by the exposure–response analysis is used to summarize the overall risk of take associated with the impact mechanisms produced by each subactivity type. The summary risk of take analysis is presented in Section 9 (*Potential Risk of Take*), which presents the risk of take associated with each subactivity type using: (1) a narrative discussion of the risk of take associated with each subactivity type by the specific associated submechanism of impact; and (2) risk of take assessment matrices that rate the risk of take resulting from each subactivity by impact mechanism and environment type. The risk of take ratings presented in the text and matrices in Section 9 are based upon the rating criteria defined in Table 6-3.

Based on the identification of impacts and risk of take analysis, additional recommendations (e.g., conservation, management, protection, and BMPs) for minimizing or mitigating project impacts were developed. (These are presented in Section 11 [*Habitat Protection, Conservation, Mitigation, and Management Strategies*].)

Table 6-3. Definitions of the terminology used for risk of take determinations in this white paper.

Risk of Take Code	Potential for Take	Definition
H	High	Stressor exposure is likely to occur with high likelihood of individual take in the form of direct mortality, injury, and/or direct or indirect effects on long-term survival, growth, and fitness potential due to long-term or permanent alteration of habitat capacity or characteristics. Likely to equate to a Likely to Adversely Affect (LTAA) finding.
M	Moderate	Stressor exposure is likely to occur, causing take in the form of direct or indirect effects potentially leading to reductions in individual survival, growth, and fitness due to short-term to intermediate-term alteration of habitat characteristics. May equate to an LTAA or a Not Likely to Adversely Affect (NLTAA) finding depending on specific circumstances.
L	Low	Stressor exposure is likely to occur, causing take in the form of temporary disturbance and minor behavioral alteration. Likely to equate to an NLTAA finding.
I	Insignificant	Stressor exposure may potentially occur, but the likelihood is discountable and/or the effects of stressor exposure are insignificant. Likely to equate to an NLTAA finding.
N	No Risk	No risk of take ratings apply to species with no likelihood of stressor exposure because they do not occur in habitats that are suitable for the subactivity type in question, or the impact mechanisms caused by the subactivity type will not produce environmental stressors.
?	Unknown	Unknown risk of take ratings apply to cases where insufficient data are available to determine the probability of exposure or to assess stressor response.

LTAA = Likely to Adversely Affect.
NLTAA = Not Likely to Adversely Affect.

7.0 Direct and Indirect Impacts

The alteration of flow regimes (i.e., flow rates and flow variability) is often identified as one of the most serious threats to sustaining the ecological functioning of rivers, floodplains, and estuaries (Bunn and Arthington 2002). In a recent review, Bunn and Arthington (2002) identify four principles that define how flow regime can influence aquatic biodiversity:

- Flow is a major determinant of habitat in streams and rivers, which in turn will influence biotic composition and diversity.
- Aquatic organisms have evolved life-history strategies in direct response to the natural flow regimes.
- Maintaining natural patterns of longitudinal and lateral connectivity is essential to the viability of populations of aquatic species.
- The invasion and success of exotic species will increase with alterations in the natural flow regime (including the inter-basin transfer of water).

Each of the six subactivity types identified and considered in this white paper under flow control structures (see Section 4 [*Hydraulic Project Description*) will alter the natural flow regime and therefore affect species, habitats, and ecological processes as described by the principles above. This section reviews and synthesizes what is known about the potential impacts of each of the identified impact mechanisms (as identified in Section 6 [*Conceptual Framework of Assessing Impacts*]) on HCP species. It reflects the literature findings on both direct and indirect impacts of activities associated with flow control structures. The exposure-response matrices (included in Appendix A) provide a synthesis of each of the HCP species (or species groupings). The matrix structure differs slightly from the text below to provide consistency with the Exposure-Response Matrices presented in other WDFW white papers [e.g., (Herrera 2007c)]. The matrix structure differs slightly from the text primarily in that the matrices define impacts with respect to different environments (i.e., riverine, lacustrine, marine), whereas the text below discusses these together.

7.1 Dams

For this white paper, dams are defined as structures built within a stream to control flow for flood control or navigation, divert flow for irrigation, or to utilize flow for the generation of hydropower. This section addresses the direct and indirect impacts of dams on HCP fish and invertebrate species, their habitats, and ecological processes. Also included in this discussion is information on how dam removal may affect fish and invertebrates and their habitats. Dam removal is becoming more common as dams come up for relicensing; therefore, it is important to address the potential impacts from this activity. Although dams are primarily located on rivers, they have indirect impacts on downstream environments such as estuaries and coastlines. Where appropriate, impacts on these environments are also presented. Furthermore, this section focuses primarily on large dams. Smaller dam structures, where flow over the structure is common, act similar to weirs, as discussed in detail in Section 7.2 (*Weirs*).

The mechanisms of impact from dams are categorized into six general impact pathways or mechanisms: construction and maintenance activities, hydraulic and geomorphic modifications, ecosystem fragmentation, riparian vegetation modifications, aquatic vegetation modifications, and water quality modifications.

In addition, another potential impact of dams (and several other flow control structures) is related to the loss of opportunities. "Lost-opportunity impacts" result from projects that adversely alter natural fluvial processes important to the ongoing creation of fish and wildlife habitats (WDFW 2003). The following quote from WDFW (2003) provides a definition of lost-opportunity impacts:

> "*Preventing a channel from naturally migrating across the floodplain usually eliminates sources of woody debris, sediment and side channels; these losses are defined as "lost opportunities." Natural channels evolve over time and migrate across their floodplains. When a channel naturally moves to a new alignment, it leaves behind vital habitat, such as floodplain sloughs and side channels. Those habitats have a finite productive longevity, some likely less than 20 years. If the natural fluvial processes of a stream are restricted or interrupted, these side-channel habitats will diminish in productivity and will not be replaced. These habitats cannot be mitigated by the design of a project. They are lost when a channel is fixed in a specific location, regardless of the bank-protection technique. Lost-opportunity impacts last as long as channel migration is halted.*"

Likewise, flow control structures can impose lost-opportunity costs through a number of pathways. One pathway is analogous to the example provided above. Certain types of flow control structures may pose lingering ecological effects when they are not designed properly to account for channel movement or to allow sediment and wood transport. An equally important type of lost-opportunity cost occurs when the structures do not provide adequate passage for all fish and invertebrate species or life-history stages.

For dams, lost opportunity impacts will occur primarily as a result of hydraulic and geomorphic modifications and ecosystem fragmentation.

7.1.1 Construction and Maintenance Activities

Construction and maintenance activities can have direct and indirect effects on HCP species. Activities associated with dam construction, maintenance, repair, or removal pose the risk of increasing underwater noise levels, increasing suspended solids, removing or disturbing aquatic and riparian vegetation, disturbing banks and shorelines, and releasing toxic substances from construction materials and/or construction equipment to fresh and marine waters. Construction and maintenance activities may also involve filling and dredging, which can entrain organisms or permanently remove habitat for burrowing and benthic animals. In summary, construction and maintenance activity impacts include a range of activities that are short-lived but intensive; these activities are required to build dams as well as to provide or maintain access to their infrastructure. The six submechanisms of impact identified for analysis in this white paper include: (1) elevated underwater noise (from pile driving and non-pile driving activities); (2) bank/channel/shoreline disturbance; (3) contamination from chemical and fuel spills; (4) dewatering, flow bypass, and fish handling; (5) channel rewatering; and (6) construction and maintenance dredging. Direct and indirect effects on fish and invertebrates are summarized below for each of these submechanisms, based on the literature review and subsequent analysis. Note: some of the information presented in this white paper is reproduced from previously prepared white papers addressing the effects of dredging and overwater structures in marine environments (Nightingale and Simenstad 2001a, 2001b).

7.1.1.1 Elevated Underwater Noise

Projects permitted under the WDFW HPA program can produce underwater noise through a variety of mechanisms. These mechanisms include construction-related noise impacts from impulsive sources (i.e., short duration, high-intensity noise from sources such as pile driving or materials placement), as well as continuous noise sources (e.g., vessel or equipment operation). The discussion presented in this section provides the noise-related analytical basis for the development of the exposure–response matrices (Appendix A) and the risk of take analysis (Section 9).

This section summarizes existing information on sources of underwater noise, how underwater noise is characterized, existing and proposed effects thresholds, and the magnitude of noise stressors associated with typical project construction and maintenance activities. This discussion is derived in part from a summary of current science on the subject developed by WSDOT (2006).

7.1.1.1.1 Measurement of Underwater Noise

Underwater sound levels are measured with a hydrophone, or underwater microphone, which converts sound pressure to voltage, which is then converted back to pressure, expressed in pascals (Pa), pounds per square inch (psi), or decibel (dB) units. Derivatives of dB units are most commonly used to describe the magnitude of sound pressure produced by an underwater noise source, with the two most commonly used measurements being the instantaneous peak sound pressure level (dB_{PEAK}) and the root mean square (dB_{RMS}) pressure level during the impulse, referenced to 1 micropascal (re: 1µPa) (Urick 1983). The dB_{PEAK} measure represents the instantaneous maximum sound pressure observed during each pulse. The RMS level represents the square root of the total sound pressure energy divided by the impulse duration, which measures the total sound pressure level produced by an impulsive source. The majority of literature uses dB_{PEAK} sound pressure to evaluate potential injury to fish. However, USFWS and NOAA Fisheries have used both dB_{PEAK} (for injury) and dB_{RMS} (for behavioral effects) threshold values to evaluate adverse effects on fish, marine mammals, and diving birds (Stadler 2007; Teachout 2007; WSDOT 2006). dB_{RMS} values are used to define disturbance thresholds in fish species, meaning the sound pressure level at which fish noticeably alter their behavior in response to the stimulus (e.g., through avoidance or a "startle" response). dB_{PEAK} values are used to define injury thresholds in salmonids, meaning the sound pressure level at which injury from barotraumas may occur (i.e., physical damage to body tissues caused by a sharp pressure gradient between a gas or fluid-filled space inside the body and the surrounding gas or liquid). Unless otherwise noted, all sound pressure levels cited herein are in dB_{PEAK} or dB_{RMS} re: 1µPa.

Noise behaves in much the same way in air and in water, attenuating gradually over distance as the receptor moves away from the noise source. However, underwater sound exhibits a range of behaviors in response to environmental variables (Urick 1983). For example, sound waves bend upward when propagated upstream into currents and downward when propagated downstream in the direction of currents. Sound waves also bend toward colder, denser water. Haloclines and other forms of stratification can also influence how sound travels. Noise shadows created by bottom topography and intervening land masses or artificial structures can, under certain circumstances, block the transmission of underwater sound waves. In freshwater systems, sound propagation is often influenced by depth and channel morphology. Underwater noise does not transmit as effectively when water depths are less than 3 feet due to the amplitude of the sound pressure wave (Urick 1983). Because underwater sound does not travel around obstructions, bends in a river or large changes in gradient will truncate sound propagation. This will limit the physical extent of noise related impacts.

Underwater noise attenuation, or transmission loss, is the reduction of the intensity of the acoustic pressure wave as it propagates, or spreads, outward from a source. Propagation can be categorized using two models: spherical spreading and cylindrical spreading. Spherical (free-field) spreading occurs when the source is free to expand with no refraction or reflection from boundaries (e.g., the bottom or the water surface). Cylindrical spreading applies when sound energy spreads outward in a cylindrical fashion bounded by the sediment and water surface. Because neither model applies perfectly in any given situation, most experts agree that a combination of the two best describes sound propagation in real-world conditions (Vagle 2003).

Currently, USFWS and NOAA Fisheries are using a practical spreading loss calculation, which accommodates this view (Stadler 2007; Teachout 2007). This formula accommodates some of the complexity of underwater noise behavior, but it does not account for a number of other factors that can significantly affect sound propagation. For example, decreasing temperature with depth can create significant shadow zones where actual sound pressure levels can be as much as 30 dB lower than calculated because sound bends toward the colder deeper water (Urick 1983). Haloclines, current mixing, water depth, acoustic wavelength, sound flanking (i.e., sound transmission through bottom sediments), and the reflective properties of the surface and the bottom can all influence sound propagation in ways that are difficult to predict.

Given these complexities, characterizing underwater sound propagation inherently involves a large amount of uncertainty. An alternative calculation approach, not used by USFWS or NOAA Fisheries, known as the Nedwell model, indirectly accounts for some of these factors. Nedwell and Edwards (2002) and Nedwell et al. (2003) measured underwater sound levels associated with pile driving close to and at distance from the source in a number of projects in English rivers. They found that the standard geometric transmission loss formula used in the practical spreading loss model did not fit well to the data, most likely because it does not account for the aforementioned factors that affect sound propagation. They developed an alternative model based on a manufactured formula that produced the best fit to sound attenuation rates measured in the field. This model thereby accounts for uncharacterized factors that affect noise attenuation, but does not explicitly identify each factor or its specific effects. Because there is considerable uncertainty regarding how to model the many factors affecting underwater noise propagation, and this would require site-specific information that cannot practically be obtained in many instances, the Services (i.e., USFWS and NOAA Fisheries) use the more conservative practical spreading loss model in ESA consultations (Stadler 2007; Teachout 2007).

7.1.1.1.2 Project-Related Noise Sources

The underwater noise produced by an HPA-permitted project, either during construction or operation, is defined by the magnitude and duration of underwater noise above ambient noise levels. The action area for underwater noise effects in ESA consultations is defined by the distance required to attenuate construction noise levels to ambient levels, as calculated using the practical spreading loss calculation or other appropriate formula provided in evolving guidance from USFWS and NOAA Fisheries on this subject.

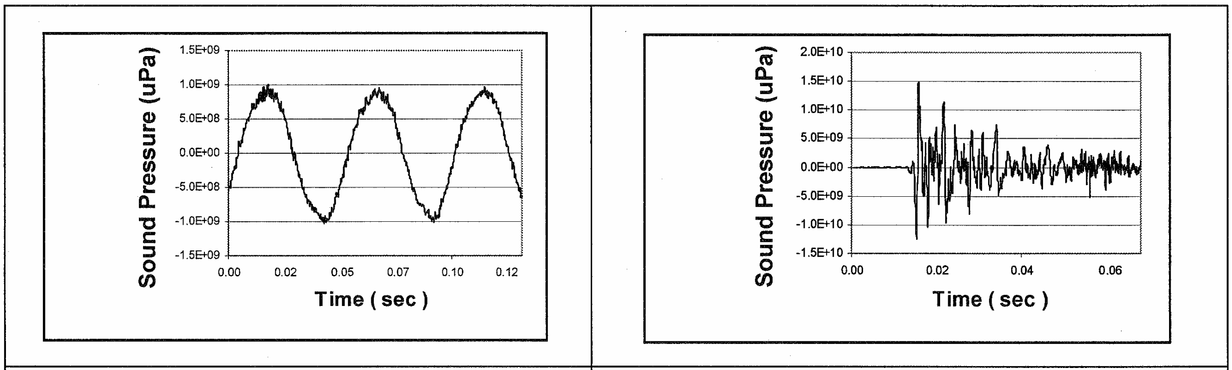
Although there are many sources of noise in the underwater environment, the following are typical sources of underwater noise associated with HPA-permitted projects:

- Project construction: Equipment operation and materials placement
- Project maintenance: Vessel operation, equipment operation.

Most sources of underwater noise potentially resulting from materials placement during HPA-permitted projects have received relatively little direct study. Of the potential sources of construction-related noise, pile driving has received the most scrutiny because it produces the highest intensity stressors capable of causing noise-related injury. Other sources of underwater noise, such as dumping of large rock or underwater tool use, have received less study.

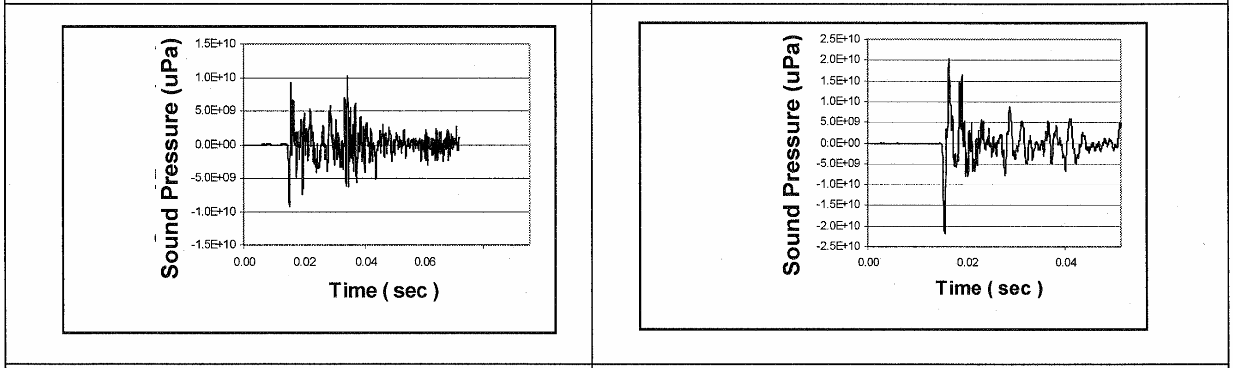
<u>Materials Placement (Pile Driving)</u>

Two major types of <u>pile-driving hammers</u> are commonly used, vibratory hammers and impact hammers. There are four kinds of <u>impact hammers:</u> diesel, air or steam driven, hydraulic, and drop hammer (typically used for smaller timber piles). Vibratory hammers produce a more rounded sound pressure wave with a slower rise time. In contrast, impact hammers produce sharp sound pressure waves with rapid rise times, the equivalent of a punch versus a push in comparison to vibratory hammers. The sharp sound pressure waves associated with impact hammers represent a rapid change in water pressure level with greater potential to cause injury or mortality in fish and invertebrates. Because the more rounded sound pressure wave produced by vibratory hammers produces a slower increase in pressure, the potential for injury and mortality is reduced. (Note that while vibratory hammers are often used to drive piles to depth, load-bearing piles must be “proofed” with some form of impact hammer to establish structural integrity.) The changes in pressure waveform generated by these different types of hammers are pictured in Figure 7-1.



Vibratory Hammer wave form—set duration

Air or steam-driven hammer wave form—single strike



Diesel hammer wave form—single strike

Hydraulic hammer wave form—single strike

Figure 7-1. Changes in pressure or underwater waveform generated by hammer type (WSDOT 2006).

Piling composition also influences the nature and magnitude of underwater noise produced during pile driving. Driven piles are typically composed of one of three basic material types: timber, concrete, or steel (although other special materials such as plastic may be used). Steel piles are often used as casings for pouring concrete piles. Noise levels associated with each of these types of piles are summarized in Table 7-1. Reference noise levels are denoted in both dB_{PEAK} and dB_{RMS} values, at the specified measurement reference distance.

Table 7-1. Reference noise levels, by structure type.

Material Type and Size	Impact Hammer Type	Reference Noise Levels[a]		Environment Type	Source
		dB_{PEAK}	dB_{RMS}		
12-inch timber	Drop	177 @ 10 m	165 @ 10 m	Marine	(Illingworth and Rodkin 2001)
24-inch concrete piles	Unspecified	188 @ 10 m	173 @ 10 m	Unspecified	[DesJardin 2003, personal communication cited by WSDOT (2006)], (Hastings and Popper 2005)
Steel H-piles	Diesel	190 @ 10 m	175 @ 10 m	Marine	(Hastings and Popper 2005; Illingworth and Rodkin 2001)
12-inch steel piles	Diesel	190 @ 10 m	190 @ 10 m	Marine	(Illingworth and Rodkin 2001)
14-inch steel piles	Hydraulic	195 @ 30 m	180 @ 30 m	Marine	(Reyff et al. 2003)
16-inch steel piles	Diesel	198 @ 10 m	187 @ 9 m	Freshwater	(Laughlin 2004)
24-inch steel piles	Diesel	217 @ 10 m	203 @ 10 m	Unspecified	(WSDOT 2006)
24-inch steel piles	Diesel	217 @ 10 m	203 @ 10 m	Unspecified	(Hastings and Popper 2005)
30-inch steel piles	Diesel	208 @ 10 m	192 @10 m	Marine	(Hastings and Popper 2005)
66-inch steel piles	Hydraulic	210 @ 10 m	195 @ 10 m	Marine	(Reyff et al. 2003)
96-inch steel piles	Hydraulic	220 @ 10 m	205 @ 10 m	Marine	(Reyff et al. 2003)
126-inch steel piles	Hydraulic	191 @ 11 m	180–206 @ 11 m	Marine	(Reyff et al. 2003)
150-inch steel piles	Hydraulic	200 @ 100 m	185 @ 100 m	Marine	(Reyff et al. 2003)

[a] Metric distances are listed as they were provided in the source material; 9 m = 29.5 ft; 10 m = 32.8 ft; 11 m = 36 ft; 30 m = 98 ft; 100 m = 328 ft.

All sound pressure values in units re: 1µPa.

Vessel/Equipment Operation and Materials Placement (Non-Pile Driving)

In comparison to pile driving, data on noise levels produced by placement of other construction-related materials is limited. For example, measured noise levels associated with work on the Friday Harbor ferry terminal (Washington) ranged between 133 dB_{peak} and 140 dB_{peak}, excluding pile driving. These noise levels were slightly higher than ambient levels, which include routine vessel traffic (Laughlin 2005). Nedwell et al. (1993) measured noise produced by underwater construction tools such as drills, grinders, and impact wrenches at 3.28 ft (1 m) from the source. When corrected for a reference distance 32.8 ft (10 m) from the source using the practical spreading loss model, the noise associated with these sources ranged from approximately 120 to 165 dB_{peak}. These data suggest that noise associated with these activities, such as in-water tool use, placement of large rock and similar material, vessel operation, and in-water operation of heavy machinery, will generally produce substantially lower noise levels than those associated with pile driving. However, other construction-related noises may generate continuous noise for longer periods, with the effect of elevating ambient noise levels or masking ambient noises in the aquatic environment that fish would ordinarily use to identify prey and predators.

This effect may be of particular concern for projects that result in changes in vessel operation or equipment use that alter ambient noise levels for longer periods (e.g., days to years). For example, vessel operation can significantly influence ambient noise levels. Large vessel engines can produce underwater sound up to 198 dB, and depth sounders can produce noise in excess of 180 dB (Buck 1995; Heathershaw et al. 2001). Hazelwood and Connelly (2005) monitored fishing vessel noise over a broad octave range from 10 Hz–40 kHz and documented noise levels ranging from 140–185 dB_{peak}, with the loudest noise occurring at the lower end of the octave range. Commercial sonar devices operating in a frequency range of 15–200 kHz can produce underwater noise ranging from 150–215 dB at maximum levels (Stocker 2002).

Ambient underwater noise levels serve as the baseline for measuring the disturbance created by project construction or maintenance. Both natural environmental noise sources and mechanical or human-generated noise contribute to the ambient or baseline noise conditions within and surrounding a project site. Therefore, these noise measurements, particularly those recorded in the vicinity of ferry terminals and other high-activity locations, are indicative of the level of noise that could be produced by project construction and operation.

Ambient noise levels have been measured in several different marine environments on the West Coast and are variable depending on a number of factors, such as site bathymetry and human activity. For example, measured ambient levels in Puget Sound are typically around 130 dB_{peak} (Laughlin 2005). However, ambient levels at the Mukilteo ferry terminal reached approximately 145 dB_{peak} in the absence of ferry traffic (WSDOT 2006). Ambient underwater noise levels measured in the vicinity of the Friday Harbor ferry terminal ranged between 131 and 136 dB_{peak} (Laughlin 2005). Carlson et al. (2005) measured the underwater baseline for the Hood Canal and found it to range from 115 to 135 dB_{RMS}. Heathershaw et al. (2001) reported open-ocean ambient noise levels to be between 74 and 100 dB_{peak} off the coast of central California. Note, however, that these ambient noise levels are typical conditions, and typical conditions can be punctuated by atypical natural events. For example, lightning strikes can produce underwater noise levels as high as 260 dB_{peak} in the immediate vicinity (Urick 1983).

Limited data are available on ambient noise levels in freshwater environments, but it is reasonable to conclude that they vary considerably based on available information. For example, high-gradient rivers, fast-flowing rivers, and large rivers and lakes with significant human activity are likely to produce more noise than lakes and slow-flowing rivers in more natural environments. Burgess and Blackwell (2003) measured ambient sounds in the Duwamish River in Seattle, Washington (averaged over 20 seconds to 5 minutes) and found the sound to vary between 110 and 130 dB continuous sound pressure sound exposure level (SEL) (SEL provides a measure of total sound pressure exposure and is expressed as dB re: $1\mu Pa^2$/second). Amoser and Ladich (2005) measured ambient noise levels in the mainstem Danube River, a smaller, fast-flowing tributary stream, a small lake, and a quiet river backwater. The river and stream represented fast-flowing habitats, the lake and backwater quiet, slow-flowing habitats. Sound behavior was complex. They found that ambient noise levels ranged from as low as 60 to as high as 120 dB_{peak} in the fast-flowing habitats, depending on the sound frequency (lower frequency sound was typically louder). Ambient noise in the slackwater habitats was considerably lower, ranging from 40–80 dB_{peak} across the frequency range (again with lower frequency sounds being loudest).

7.1.1.1.3 Direct and Indirect Effects on Fish

Most fish sense sounds, vibrations, and other displacements of water in their environment through their inner ear and with the lateral line running the length of each side of the fish and on the head. The lateral line is a mechano-sensory system that plays an indirect role in hearing through its sensitivity to pressure changes at close range. The hearing organs and lateral line system are collectively referred to as the acoustico-lateralis system. The hearing thresholds of different fish species vary depending on the structure and sensitivity of this system. Those families of fish known as hearing specialists include cyprinids (dace [e.g., Umatilla and leopard dace], minnows and carp), catastomids (suckers [e.g., mountain sucker]), and ictalurids (catfish), which collectively belong to the Ostariophysan taxonomic grouping of fishes. These fish possess a physical connection between the swim bladder and the inner ear, with the swim bladder acting as an amplifier that transforms the pressure component of sound into particle velocity component, to which the inner ear is sensitive (Moyle and Cech 1998). The hearing capacity of salmonids, on the other hand, is limited both in bandwidth and intensity threshold. The Atlantic salmon, for example, is functionally deaf at sound pressure wavelengths above 380 Hz (Hawkins and Johnston 1978). In these fish, the swim bladder does not likely enhance hearing.

Noise sources such as pile driving that produce high-intensity sound pressure waves can result in direct effects on fish ranging from effects as limited as temporary stress and behavioral avoidance, to temporary or permanent injury in multiple organ systems (including hearing, heart, kidney, swim bladder, and other vascular tissue), to direct mortality (Popper and Fay 1973; 1993). Another potential effect includes masking of existing ambient noise, reducing the ability of fish to sense predators or prey. These activities may also have indirect effects such as reducing the foraging success of these fish by affecting the distribution or viability of potential prey species. Numerous studies have examined the effects on fish associated with underwater noise and are described more fully below.

In general, injury and mortality effects from underwater noise are caused by rapid pressure changes, especially on gas-filled spaces in the body. Rapid volume changes of the swim bladder may cause it to tear, resulting in a loss of hearing sensitivity and hydrostatic control. Intense noise may also damage the tissue in hearing organs, as well as the heart, kidneys, and other highly vascular tissue. Susceptibility to injury is variable and depends on species-specific physiology, auditory injury, and auditory thresholds (Popper and Fay 1973; 1993). While species-specific data are limited, the available information indicates variable effects related to physiology, size, and age, as well as the intensity, wavelength, and duration of sound exposure.

Hardyniec and Skeen (2005) and Hastings and Popper (2005) summarized available information on the effects of pile driving-related noise on fish. Pile driving effects observed in the studies reviewed ranged broadly from brief startle responses followed by habituation to instantaneous lethal injury. The difference in effect is dependent on a number of factors, including piling material, the type and size of equipment used, and mitigation measures; site-specific depth, substrate, and water conditions; and the species, size, and life-history stage of fish exposed.

Popper et al. (2005) exposed three species of fish to high-intensity percussive sounds from a seismic air gun at sound levels ranging between 205 and 209 dB$_{\text{Peak}}$, intending to mimic exposure to pile driving. Subject species included a hearing generalist (broad whitefish), a hearing specialist (lake chub), and a species that is intermediate in hearing (northern pike). They found that the broad whitefish suffered no significant effects from noise exposure, the lake chub demonstrated a pronounced temporary threshold shift in hearing sensitivity (i.e., hearing loss), and the northern pike showed a significant temporary hearing loss but less than that of the lake chub. The hearing sensitivity of lake chub and northern pike returned to their respective normal thresholds after 18–24 hours. High-intensity sounds can also permanently damage fish hearing (Popper and Clarke 1976; Enger 1981; Cox et al. 1987).

Enger (1981) found that pulsed sound at 180 dB was sufficient to damage the hearing organs of codfish (genus *Gadus*), resulting in permanent hearing loss. Hastings (1995) found that goldfish exposed to continuous tones of 189, 192, and 204 dB$_{\text{peak}}$ at 250 Hz for 1 hour suffered permanent damage to auditory sensory cells. Injury effects may also vary depending on noise frequency and duration. Hastings et al. (1996) found destruction of sensory cells in the inner ears of oscars 4 days after exposure to continuous sound for 1 hour at 180 dB$_{\text{peak}}$ at 300 Hz. In contrast, when the two groups of the same species were exposed to continuous sound at 180 dB$_{\text{peak}}$ at 60 Hz for 1 hour, and to impulsive sound at 180 dB$_{\text{peak}}$ at 300 Hz repeatedly over 1 hour, they showed no apparent injury. Susceptibility to injury may also be life-history stage specific. Banner and Hyatt (1973) demonstrated increased mortality of sheepshead minnow eggs and embryos when exposed to broadband noise approximately 15 dB above the ambient sound level. However, hatched sheepshead minnow fry were unaffected by the same exposure.

Even in the absence of injury, noise can produce sublethal effects. Behavioral responses to sound stimuli is well established in the literature for many fish species. For example, Moore and Newman (1956) reported that the classic fright response of salmonids to instantaneous sound stimuli was the "startle" or "start" behavior, where a fish rapidly darts away from the noise source. Knudsen et al. (1992) found that in response to low-frequency (10 Hz range) sound, salmonids 1.6–2.4 in (40–60 mm) in length exhibited an initial startle response followed by habituation, while higher frequency sound caused no response even at high intensity. In a study of the effects of observed pile-driving activities on the behavior and distribution of juvenile pink and chum salmon, Feist et al. (1992) found that pile-driving operations were associated with changes in the distribution and behavior of fish schools in the vicinity. Fish schools were two-fold more abundant during normal construction days in comparison to periods when pile driving took place. Blaxter et al. (1981) found Atlantic herring to exhibit an avoidance response to both continuous pulsed sound stimuli with habituation to more continuous stimuli occurring over time, and Schwarz and Greer (1984) found similar responses on the part of Pacific herring. Sound has also been shown to affect growth rates, fat stores, and reproduction (Meier and Horseman 1977; Banner and Hyatt 1973).

Prolonged underwater noise can also reduce the sensitivity of fish to underwater noise stimuli, with potentially important effects on survival, growth, and fitness. The fish auditory system is likely one of the most important mechanisms fish use to detect and respond to prey, predators, and social interaction (Scholik and Yan 2001, 2002; Hawkins 1986; Fay 1988; Kalmijn 1988; Cox et al. 1987; Myrberg 1972; Myrberg and Riggio 1985; Wisby et al. 1964; Nelson 1965; Nelson et al. 1969; Richard 1968; Amoser and Ladich 2005). Scholik and Yan (2001) studied the auditory responses of the cyprinid fathead minnow to underwater noise levels typical of human-related activities (e.g., a 50-horsepower outboard motor). They found that prolonged exposure decreased noise sensitivity, increasing the threshold level required to elicit a disturbance response for as long as 14 days after the exposure. Amoser and Ladich (2005) reported similar findings in common carp in the Danube River, noting that auditory ability in this hearing specialist species was measurably masked in environments with higher background noise. They reported similar but far less pronounced responses in hearing generalist species such as perch. These data suggest that elevated ambient noise levels have the potential to impair hearing ability in a variety of fish species, which may in turn adversely affect the ability to detect prey and avoid predators, but that this effect is variable depending on the specific sensitivity of the species in question. Feist et al. (1992) similarly theorized that it was possible that auditory masking and habituation to loud continuous noise from machinery may decrease the ability of salmonids to detect approaching predators.

7.1.1.1.4 Direct and Indirect Effects on Invertebrates

In general, information on the effects of underwater noise on invertebrates is limited, indicating that additional research on the subject is needed. What little data are available suggest some sensitivity to intense percussive underwater noise. In a study completed by Turnpenny et al. (1994), mussels, periwinkles, amphipods, squid, scallops, and sea urchins were exposed to high air gun and slow-rise-time sounds at between 217 and 260 $\mathrm{dB_{peak}}$, analogous to extremely loud pile driving. One scallop suffered a split shell following exposure to 217 $\mathrm{dB_{peak}}$, suggesting the potential for serious injury when percussive underwater noise exceeds these levels.

No research has been identified regarding the effects of lower intensity continuous underwater noise on invertebrates. However, operational noise is typically associated with sound pressures well below levels that have been observed to cause injury in shellfish, suggesting that HCP invertebrate species might not be subject to these effects. Because HCP invertebrate species with the potential for stressor exposure are either filter feeders or grazers and are essentially non-motile, these species are unlikely to be subject to auditory masking effects that would limit the ability to sense predators and prey. Some potential may exist for disturbance-induced interruption of feeding behavior, but more research on this subject is necessary to determine this definitively and this subject is considered a data gap.

7.1.1.2 Bank/Channel/Shoreline Disturbance

Dam construction (and removal) involves heavy equipment that will repeatedly enter and exit the channel. The presence of this equipment will lead to disturbances in both riparian and aquatic vegetation. As a result, areas within the construction zone may experience removal and/or damage of riparian vegetation resulting in erosion and soil loss (USEPA 2007). Within the channel, aquatic vegetation may be removed, damaged, or scoured downstream from equipment operation. Impacts from bank/channel/shoreline disturbance will vary depending on the type and size of dam being built (or removed). However, the resultant stressors will be the same. Disturbance within the channel will result in increased suspended solids, both from bank and bed activities and from erosion and bank failure associated with equipment operation. This sediment pulse will be short term (during construction) but may be detrimental to sensitive species, particularly those species (e.g., freshwater mussels and snails) or life-history stages (e.g., salmon embryos) that cannot avoid the increased turbidity. A detailed discussion of the effects of increased suspended solids on fish and invertebrates is provided in Section 7.1.6.3 (*Altered Suspended Sediment and Turbidity*). Besides impacts from increased suspended sediment and turbidity, riparian vegetation losses can alter temperature regime, shading, allochthonous inputs, groundwater-surface water interactions, habitat complexity, and nutrient loading. These additional impacts are discussed in detail in Section 7.1.4 (*Riparian Vegetation Modifications*). Additional impacts from aquatic vegetation losses include altered autochthonous production, habitat complexity, and nutrient loading and are discussed in Section 7.1.5 (*Aquatic Vegetation Modifications*).

7.1.1.3 Contamination from Chemical and Fuel Spills

According to USEPA (2007), although sediment is the major source of pollution, the following pollutants can be associated with dam construction and maintenance activities:

- Petroleum products–fuels and lubricants, specifically gasoline, diesel oil, kerosene, lubricating oils, grease, and asphalt
- Pesticides–insecticides, herbicides, fungicides, and rodenticides
- Fertilizers
- Construction chemicals–acids, soil additives, and concrete-curing compounds
- Wastewater–aggregate wash water, herbicide wash water, concrete-curing water, core-drilling wastewater, or clean-up water from concrete mixers
- Solid wastes–paper, wood, metal, rubber, plastic, and roofing materials
- Garbage
- Sanitary wastes
- Cement
- Lime.

Contamination from the above sources can cause alterations in pH levels, nutrients, organic contaminants, and metals. The direct and indirect impacts from these contaminants on fish and invertebrates are discussed in detail in Section 7.1.6 (*Water Quality Modifications*).

7.1.1.4 Dewatering, Flow Bypass, and Fish Handling

In many cases, construction of HPA-permitted projects may require the exclusion of streamflows or even the dewatering of the work area to protect aquatic life and/or provide a suitable environment for construction. These activities have the potential to cause direct and indirect effects on HCP species. Fish exclusion and dewatering involve the placement of barriers (e.g., block nets, temporary berms, and cofferdams) around a work area and the capture and removal of fish and other aquatic life within the work area. Electrofishing is a common practice used for fish capture in freshwater environments, as is the use of minnow traps, hand nets, beach seines, and other net-based capture methods. Because electrofishing is ineffective in brackish or salt water, net-based capture methods are used in these environment types.

The direct effects of fish exclusion and dewatering include:

- Direct mortality, injury, and stress from electrical field exposure (i.e., electrofishing)
- Capture by netting, leading to direct mortality, injury, and stress
- Physical and thermal stress and possible trauma associated with handling and transfer during capture and transfer between temporary holding containers and release locations
- Direct mortality, injury, and stress from stranding and asphyxiation
- Entrainment or impingement in block nets, dewatering pumps, and bypass equipment
- Increased stress, predation exposure, and habitat competition once relocated
- Increased competition for aquatic species forced to compete with relocated animals.

Exclusion areas may also create temporary barriers to fish passage, with attendant effects on migratory fish species. For a detailed discussion on the impacts to fish and invertebrates from fish passage, see the Fish Passage White Paper (Herrera 2007a).

7.1.1.4.1 Direct and Indirect Effects

Of the various methods used for dewatering and fish handling, the majority of research has been conducted on incidental mortality and injury rates associated with electrofishing. Much of this research has focused on adult salmonids greater than 12 inches in length (Dalbey et al. 1996). The relatively few studies that have been conducted on juvenile salmonids suggest spinal injury rates lower than those observed for large fish, perhaps because juvenile fish generate less total electrical potential along a shorter body length (Dalbey et al. 1996; Sharber and Carothers 1988; Thompson et al. 1997). Electrofishing-related injury rates are variable, reflecting a range of factors from fish size and sensitivity, individual site conditions, to crew experience and the type of equipment used, with the equipment type being a particularly important factor (Dalbey et al. 1996; Dwyer and White 1997; Sharber and Carothers 1988). Electrofishing equipment typically uses continuous direct current (DC) or low-frequency pulsed DC equipment. The use of low-frequency DC (equal to or less than 30 Hz) is the recommended electrofishing method because it is associated with lower spinal injury rates (Ainslie et al. 1998; Dalbey et al. 1996; Fredenberg 1992). Even with careful selection of equipment, observed injury rates can vary. For example, one study in the Yakima River basin (McMichael et al. 1998) observed a 5.1 percent injury rate for juvenile steelhead captured using 30 Hz pulsed DC equipment. Ainslie et al. (1998) reported injury rates of 15–39 percent in juvenile rainbow trout using continuous and pulsed DC equipment, and found that while pulsed DC equipment produced injury more frequently, these injuries were less severe in nature.

It is notable that electrofishing capture typically has a low direct mortality rate, but it is reasonable to conclude that injuries induced by electrofishing could have long-term effects on survival, growth, and fitness. The few studies that have examined this question found that few juvenile salmonids die as a result of electrofishing-induced spinal injury (Ainslie et al. 1998; Dalbey et al. 1996). However, fish with more injuries demonstrated a clear decrease in growth rates, and in some cases growth was entirely arrested (Dalbey et al. 1996). In the absence of additional supporting information, it is reasonable to conclude that these same effects would affect many of the HCP fish species, but this conservative assumption may not be universally accurate. Studies of the effects of electrofishing on other fish species are more limited, but available data indicate that at least some HCP species may be less sensitive to injury-related effects. Holliman et al. (2003) exposed a threatened cyprinid (minnow) species to electrofishing techniques in the laboratory and found that the typical current and voltage parameters used to minimize adverse effects on salmonid species produced no evidence of injury. This suggests that other cyprinids such as leopard and spotted dace, lake chub, and suckers may also be less sensitive.

Beyond the effects of electrofishing, the act of capture and handling demonstrably increases physiological stress in fishes (Frisch and Anderson 2000). Primary contributing factors to handling-induced stress and death include exposure to large changes in water temperatures and dissolved oxygen conditions (caused by large differences between the capture, holding, and release environments); duration of time held out of the water; and physical trauma (e.g., due to net abrasion, squeezing, accidental dropping). Even in the absence of injury, stress induced by capture and handling can have a lingering effect on survival and productivity. One study found that handling stress impaired predator evasion in salmonids for up to 24 hours following release and caused other forms of mortality (Olla et al. 1995).

Use of a bypass system is a common means of creating exclusion areas via dewatering and flow reduction. Partial dewatering is a technique used to reduce the volume of water in the work area to make capture methods more efficient. In riverine habitats, this method is used to move fish out of affected habitats to reduce the number of individuals exposed to capture and handling stress and potential injury and mortality. Based on interviews with state fisheries agency staff, NOAA Fisheries has estimated that 50–75 percent of fish in an affected reach will volitionally move out of an affected reach when flows are reduced by 80 percent (NMFS 2006). However, volitional movement will lead to the concentration of fish in unaffected habitats, increasing competition for available space and resources.

Failure to capture and remove fish or invertebrates from work areas must also be considered. Organisms left within the exclusion area would potentially be directly exposed to stranding and asphyxiation during dewatering or, if left inundated, to mechanical injury and/or high intensity noise, turbidity and other pollutants. Many species of fish, such as salmonids and larval lamprey, are highly cryptic and can avoid being detected even when using multiple pass electrofishing because they hide in large interstices or are buried in sediments (Peterson et al. 2005; Peterson et al. 2004; Wydoski and Whitney 2003).

As noted above, research on fish injury and mortality associated with dewatering has focused predominantly on salmonids, relatively large fish species that respond well to this exclusion technique. Other species may have non-motile or cryptic life-history stages (e.g., lamprey ammocoetes buried in fine sediments) or life-history stages that cannot easily move to adjust to changes in flow or are not easily captured and relocated (e.g., adhesive eggs of eulachon, juvenile rockfish, and lingcod). In freshwater environments, examples of species and life-history stages that are sensitive to dewatering impacts include incubating salmonid eggs and alevins, lamprey ammocoetes, the adhesive eggs of eulachon, sturgeon, and other species. These life-history stages are relatively immobile and also difficult to capture and relocate efficiently. Therefore, they face a higher likelihood of exposure to stranding or entrainment in dewatering pumps, which would be expected to lead to mortality. In marine environments, the larval and juvenile life-history stages of rockfish, lingcod, Pacific cod, hake, pollock, herring, smelt, and sand lance are similarly immobile and difficult to capture. Therefore, it is reasonable to conclude that they would be vulnerable to similar effects.

HCP invertebrate species demonstrate different sensitivity to the effects of dewatering and relocation than fish, with many species being relatively insensitive to the effects of handling, at least during adult life-history stages. For example, Krueger et al. (2007) studied the effects of suction dredge entrainment on adult western ridged and western pearlshell mussels in the Similkameen River and found no evidence of mortality or significant injury. Suction dredge entrainment is expected to be a more traumatic stressor than removal and relocation by hand. These findings suggest that careful handling would be unlikely to cause injury. However, the authors cautioned that these findings were limited to adult mussels, and the potential for injury and mortality in juveniles remains unknown.

The sensitivity of other HCP invertebrate species, such as giant Columbia River limpet and great Columbia River spire snail, is somewhat less certain. Adults may be easily removed and relocated during dewatering, but juveniles and eggs may be difficult to locate and remove effectively. This suggests the potential for mortality from stranding. Failure to locate and remove small or cryptic invertebrate species or life-history stages may result in stranding or concentrated exposure to other stressors within the exclusion area. Stranding caused by operational water level fluctuations was associated with mass mortality of California floater and western ridged mussels in Snake River reservoir impoundments (Nedeau et al. 2005).

While handling-related injury and mortality are relatively unlikely, relocation may lead to notable nonlethal effects. For example, scattering of closely packed groups of adult mussels may affect reproductive success if mussels are scattered outside a certain proximity. Because female freshwater mussels filter male gametes from the water column, successful fertilization is density dependent (Downing et al. 1993).

7.1.1.5 Channel Rewatering

Installation, operation, and removal of a stream bypass system to rewater a channel can increase turbidity. The in-water installation and removal work poses the highest risk of disturbing the stream bank and substrate, thereby resuspending sediments and increasing turbidity. Fish may experience short-term, adverse effects as a result of increased turbidity. The effects of increased turbidity during rewatering are discussed in Section 7.1.6.3 (*Altered Suspended Sediments and Turbidity*).

7.1.1.6 Construction and Maintenance Dredging

Construction or maintenance dredging converts shallower subtidal habitats to deeper subtidal habitats or is used to reduce the amount of accumulated sediment behind dams. Sediment accumulation behind dams can impede navigation through rivers where dams and other flow control structures (e.g., dikes and levees) are located.

7.1.1.6.1 Impact Mechanisms from Construction and Maintenance Dredging

Dredging affects fish and invertebrates by several different means, the most significant being the alteration of bathymetry, entrainment of benthic organisms, and turbidity and resuspension of contaminated sediments. These stressors are discussed below.

Altered Bathymetry and Substrate Composition

Large channel deepening projects can alter ecological relationships through the change of freshwater inflow, tidal circulation, estuarine flushing, and freshwater and saltwater mixing (Sherwood et al. 1990). Miller et al. (1990) reported that only through comprehensive areal surveys over a minimum of four seasons before dredging, with follow-up surveys after dredging, can the impacts of channel deepening on aquatic resources be determined. In a comparison of catch between dredged and undredged areas in the Port of Everett's public marina, Pentec (1991) found catches of fish to be higher before dredging than after.

Depending on site-specific characteristics, maintenance dredging may occur annually or at intervals of 10 years or longer. These different dredging timelines represent different disturbance regimes both in terms of the ability of the benthos to recolonize after disturbance and the magnitude of benthic productivity affected by dredging. In a literature review report on dredge and disposal effects, Morton (1977) reported the range of effects on invertebrate communities to be from negligible to severe, with impacts ranging from short to long term. In general, this literature review found that short-term, small-scale dredging and dredge disposal projects affected benthic communities less than long-term, large-scale projects. This is likely due to the fact that benthic communities are more likely (and quicker) to recover from short-term, less intense, small-scale disturbances than from large-scale and intense disturbances lasting long period of times (Dernie et al. 2002; Guerra-Garcia et al. 2003). For example, in experiments conducted in sheltered sand flat habitat, the recovery of the benthic community from lower intensity disturbance (sediment was removed to a depth of 3.9 inches [10 cm]) occurred within 64 days of the disturbance, whereas recovery after higher intensity disturbance (sediment removed to a 7.9-inch [20-cm] depth) did not occur until 208 days postdisturbance at this site (Dernie et al. 2002).

In a study to evaluate the effects of dredged material disposal on biological communities, Hinton et al. (1992) reported a significant increase in benthic invertebrate densities at a disposal site between June 1989 (pre-disposal) and June 1990 (post-disposal). Recolonization could have occurred by invertebrates burrowing up through newly deposited sediments or recruitment from surrounding areas (Richardson et al. 1977).

Entrainment

Entrainment occurs when an organism is trapped in the uptake of sediments and water removed by dredging machinery (Reine and Clarke 1998). Benthic infauna are particularly vulnerable to being entrained by dredging uptake, but mobile epibenthic and demersal organisms such as burrowing shrimp, crabs, and fish also can be susceptible to entrainment. Entrainment rates are usually described by the number of organisms entrained per cubic yard (cy) of dredged sediment (Armstrong et al. 1982).

Demersal fish, such as sand lance, sculpins, and sticklebacks, as well as lamprey ammocoetes likely have the highest rates of entrainment as they have life-history strategies of burrowing or hiding in the bottom substrate. Larval fish with little or no swimming capacity to avoid direct dredge impacts are also at significant risk of entrainment in dredge sites. Armstrong et al. (1982) found that larger fish were not necessarily able to avoid the hopper dredge, with the largest specimen being a 9.2-in (234-mm) tomcod.

Larger fish may also be susceptible to entrainment. Armstrong et al. (1982) found that larger fish were not necessarily able to avoid the hopper dredge, with the largest specimen being a 9.2-in (234-mm) tomcod. Tests of excluders mounted on the draghead of a hopper dredge showed that 66 percent fewer fishes (mostly flatfish and gunnels in the study) could be saved from entrainment through use of the device (Shaw 1996).

Buell (1992) found entrainment of juvenile white sturgeon (11.8–19.6 in [300–500 mm]) at a rate of 0.015 fish/cy. In another study, juvenile salmonids and eulachons were the dominant entrained taxa due to the dredge location in a constricted waterway, making it more difficult for salmonids to avoid the dredge operation (Larson and Moehl 1990; McGraw and Armstrong 1990).

Entrained bivalve larvae, such as larval oysters, are assumed to suffer 100 percent mortality by sediment smothering, anoxia, starvation, or desiccation even without direct mechanical impacts from pumping. In Chesapeake Bay, concern for oyster larvae entrainment resulted in the development of a population model using conservative temporal and spatial distributions. Using the population model, entrainment was found to have minimal negative effect on the population as the entrainment rate was calculated to range between 0.005 and 0.3 percent of the local population. Lunz (1985) concluded that this represented no significant impact as the dredge entrained only a small fraction of the total water volume flowing past the dredge. Many species, particularly marine fish and invertebrates, have planktonic larval life-history stages that suffer naturally high mortality rates (in some cases exceeding 99 percent). Therefore, the potential mortality from entrainment is insignificant in comparison (Lunz 1985).

Increased Suspended Sediment

Dredging activities are directly related to increases in suspended sediment and turbidity. In addition, a cumulative effect on HCP species will occur if these increased suspended sediments are contaminated. Many different types of pollutants are known to adsorb to sediments (Murakami and Takeishi 1977), so construction and maintenance in areas with sediment pollution would contribute to the resuspension of contaminated sediments. Numerical modeling simulations of dredging-related suspended-sediment plume dynamics are currently being developed by the U.S. Army Corps of Engineers under the Dredging Operations and Environmental Research Program (USACE 2007). The existing data indicate that responses to suspended sediments are highly species-specific, with some species having lethal effects at several hundred parts per million (ppm) in 24 hours and others having no effect at concentrations above 10,000 ppm for 7 days. Studies on east coast species have identified lethal concentration levels, and Newcombe and Jensen (1996) have developed a predictive model for defining lethal and sublethal fish injury threshold levels for suspended solids concentrations. However, threshold studies for the temporary impacts of suspended sediment levels specific to aquatic environments in the Pacific Northwest are lacking.

7.1.1.6.2 Direct and Indirect Effects

Dredging activities result in short-term direct effects, including entrainment and potential mortality; periodic removal of potentially suitable habitats for fish and invertebrates; alteration of water circulation and subsequent nutrient, prey, and habitat availability; and increased turbidity and potential resuspension of contaminants. The direct and indirect effects of suspended sediment on fish and invertebrates are summarized in Section 7.1.6.3 (*Altered Suspended Sediments and Turbidity*) and Section 7.1.6.4 (*Increases in Contaminated Sediment*).

In addition, long-term and food web indirect effects can occur, such as reconfiguration of the benthos and the availability of nutrient and prey resources. Resulting impacts, as described in detail in Section 7.1.6 (*Water Quality Modifications*), Section 7.1.5 (*Aquatic Vegetation Modifications*), and Section 7.1.2 (*Hydraulic and Geomorphic Modifications*), include mortality, injury, decreased foraging opportunity, decreased survival, decreased growth and fitness, and physiological and behavioral responses. Deposition of dredged spoils can bury existing habitats and benthic organisms, resulting in a similar suite of impacts. Research has shown potential increases in densities of invertebrates at dredge disposal sites (Hinton et al. 1992).

Direct effects associated with dredging entrainment include injury or mortality of fish and invertebrate species. Indirect effects include alteration of food web interactions that could affect overall species survival or fitness.

7.1.2 Hydraulic and Geomorphic Modifications

River hydrology includes the movement of water in the stream, the movement of hyporheic groundwater to the stream, and the movement of surface water across land to the stream. It also includes the tidal delta hydrology and the river's exchange of marine and fresh water. Changes to riverine hydrology that reduce or increase the flow of water to the river alter the suitability of habitats within the river. During low-flow periods, alterations to hydrology can result in previously wetted areas going dry, thereby eliminating habitat area for aquatic organisms. Hydrologic alterations that increase overland surface water flow can, on the other hand, increase flooding and scour substrates.

Riverine hydraulics determine the nature as well as the distribution and deposition of, sediments and other materials along the path of the river's unidirectional movement toward lower elevations, also known as the river continuum. Fishes and invertebrates depend upon the diversity of habitats created by those hydraulic forces that result in the distribution of diverse sediments along the river continuum (Montgomery et al. 1999). HCP species, such as sturgeon, char, bull trout, salmonids, and freshwater mussels, depend on particular riverine sediment types and habitats. If flows become too strong, reaches of rivers can be made impassable to various fish species or life-history stages, or unsuitable for invertebrates. The reproduction, growth, and survival of these HCP species depend on particular hydraulic regimes to maintain suitable habitats. Alterations to river hydraulics that change the flow of water and the ability of the water to move sediments and nutrients can have direct and indirect effects on HCP species. Projects that alter riverine hydrology can also have direct and indirect effects on HCP species.

7.1.2.1 Submechanisms of Impact

The presence of a dam dramatically influences the hydraulic and geomorphic properties of a riverine system. Five submechanisms of impact associated with hydraulic and geomorphic modifications have been identified for analysis in this white paper: (1) altered flow regime, (2) altered sediment transport, (3) altered channel geometry, (4) altered substrate composition; and (5) altered groundwater–surface water interactions. Direct and indirect effects on fish and invertebrates are summarized below for each of these submechanisms based on the literature review and subsequent analysis. Some impacts, such as altered sediment transport, will impact estuarine and coastal ecosystems as well. These environments are addressed where applicable.

If a dam is located close to coastal and estuarine areas, it may affect the hydraulics of nearshore environments as well. These modifications include altered wave energy, altered current velocities, and altered nearshore circulation. Details of the hydraulic impacts in marine environments from flow control structures are discussed in detail in Section 7.4.2.1 (*Submerged Outfalls*).

7.1.2.1.1 Altered Flow Regime

In this white paper, altered flow regime refers to changes in flow variability, flow rate, or flow velocity associated with each flow control structure subactivity type. A dam will directly alter the flow regime on a stream or river because its presence represents a barrier to flow. Altered flow variability is directly related to changes in flow rate and is discussed in concert with these changes.

<u>Altered Flow Variability</u>

Dams tend to reduce peak flows and increase base flows (Magilligan and Nislow 2005), especially for systems where dams are used for hydropower generation. Flow variability is changed from a natural fluctuation to one based on human needs. The changes in flow variability translate into changes in daily high and low water, which can alter flooding and inundation of side channels and floodplains, thereby affecting habitat connectivity. In addition, altered flow variability can dewater floodplain habitat and strand fish and invertebrate species. For example, a drawdown of the Lower Granite Reservoir on the lower Snake River killed many California floaters, western floaters, and western ridged mussels (Nedeau et al. 2005). In addition, freshwater mussels are known to migrate to avoid receding waters and can be vulnerable to predators during this time (Nedeau et al. 2005). If dewatering occurs for long time periods, mussels may bury themselves during dewatering, but there is a risk of mortality if waters do not return to normal levels before the mussels overheat. From these examples, it has been shown that disconnection of a river from its floodplain from alterations in flow variability can have direct and indirect impacts on fish and invertebrate species. These impacts are discussed in more detail in Section 7.1.3.1.2 (*Altered River-Floodplain Connectivity*).

The reduction in flow variability can also contribute to changes in species composition. High flows, which can displace organisms downstream, help maintain biodiversity through natural flow variability. When stable flows persist in the presence of a dam, organisms adapted to stable flows dominate and diversity will be reduced (Bednarek 2001). In contrast, dam removals have been shown to increase species diversity by restoring the natural flow variability. A dam removal on the Chipola River in Florida increased fish diversity downstream from 34 to 61 species (Hill et al. 1993). Alteration of species interactions and diversity will have an impact on fish and invertebrates and is discussed in more detail in Section 7.1.3.1.5 (*Altered Community Composition*).

Intermittent flooding and draining are needed for the regeneration of riparian forests. In the presence of dams, a loss of flooding reduces forest productivity, suppresses tree growth, and increases tree mortality (Kozlowski 2002). In addition, upstream flooding from reservoir inundation kills trees and seed sources, resulting in inadequate seed supplies for downstream forests (Kozlowski 2002). Alteration of riparian vegetation will have an impact on fish and invertebrates, as discussed in detail in Section 7.1.4 (*Riparian Vegetation Modifications*).

Alteration of flow can have impacts far downstream. Reduced freshwater flows can affect tidal mixing and translate into impacts on marine species. Migration patterns, spawning habitat, and species diversity for adult and larval stages of fish and invertebrates are affected by the presence of dams upstream (Drinkwater and Frank 1994). Changes in tidal surges will particularly impact weak swimming fish or early life-history stages that rely on swimming with tidal flows during migration upstream or downstream during spring high flows (Dadswell 1996; Oullet and Dodson 1985).

<u>Altered Flow Velocities</u>

Inherent in altered flow variability is the change in flow velocities. During times of water release, velocities downstream can become quite large; however, when water is held back, velocities downstream are depressed. Fish and invertebrates inhabiting riverine environments require certain flow velocities for spawning, rearing, migration, and foraging. For example, Chinook salmon tolerate velocities up to 49.9 ft/sec (15.2 m/sec) (Johnson et al. 2003) during migration, whereas Pacific lamprey seek out slower velocities (0–0.33 ft/sec) for rearing (Stone and Barndt 2005). Optimal velocities for spawning habitat for mountain suckers in Lost Creek, Utah, are 2.4–7.9 in (6–20 cm/sec) (Wydoski and Wydoski 2002), whereas Columbia River white sturgeon require low velocities (~2.6 ft/sec [0.8 m/sec]) for spawning (Paragamian et al. 2001) and use sand dunes for cover from high velocities (Young and Scarnecchia 2005).

Flow velocities also influence swimming activity and respiration in fish species. High flows below Hells Canyon Dam on the Snake River caused increased swimming activity and subsequently higher O_2 consumption, leading to suppressed movement in white sturgeon (Geist et al. 2005). The study suggested that high flows and velocities, even of short duration, can restrict the movement of juvenile white sturgeon; however, these increases may not cause an increase in energy expenditure due to the adaptation of white sturgeon to high-flow environments. For other HCP species that prefer slower velocities (e.g., Pacific lamprey) high velocities caused by dam releases may be more prohibitive.

Increased flow velocities during water releases can also cause fish species to rest in areas of slower moving water in order to recover from increased activity. This behavior can result in unsuccessful recruitment from delayed migration upstream for anadromous species (e.g., salmonids, sturgeon, lamprey), or increased predation from holding in slow pools downstream of dams and high-velocity reaches. Impacts related to delayed migration and increased predation are discussed in more detail in Section 7.1.3 (*Ecosystem Fragmentation*) and the Fish Passage white paper (Herrera 2007a).

Changes in flow velocities may also significantly alter sediment transport. The presence of a dam slows river water upstream, causing increased sedimentation in the impoundment behind the dam. Downstream, increased velocities from water releases can scour bed material and benthic organisms (Camargo and Voelz 1998). These changes in natural flow velocity alter natural sediment transport in the reach and are discussed in more detail in Section 7.1.2.1.3 (*Altered Sediment Transport*).

7.1.2.1.2 Altered Channel Geometry

Dams dramatically alter flow in a river, which changes channel geometry. For this white paper, channel geometry refers to width, depth, slope, and roughness of the channel. It is well known that a dam causes changes in channel width and depth. Upstream of a dam, both depth and width increase; downstream, the average depth and width decrease (Tiemann et al. 2004). However, altered sediment transport can also increase erosion downstream, widen the channel, and reduce channel roughness (Assani and Petit 2004). This change in erosion and deposition is discussed in more detail in Section 7.1.2.1.3 (*Altered Sediment Transport*). Furthermore, flow velocity in a channel is proportional to the hydraulic radius (the cross-sectional area of the channel divided by the wetted perimeter) and inversely proportional to roughness (Leopold et al. 1964). Therefore, changes in flow velocity ultimately change the channel geometry. Finally, altered depth and width downstream of a dam disconnect the river from its floodplain and side channel habitats, potentially reducing habitat accessibility and increasing the stranding of aquatic species.

7.1.2.1.3 Altered Sediment Transport

Sediment transport is defined as the movement of sediment and refers to alterations in sediment source(s). Therefore, changes in aggradation (raising) and incision (lowering) of the channel are addressed in this section. Impacts from changes in substrate composition are discussed in Section 7.1.2.1.4 (*Altered Substrate Composition*).

Dams modify the sediment available to species for spawning and rearing by blocking the contribution of sediments from upland or upstream source areas. As water velocities slow upstream of dams and as water enters the impounded area, sediment settles out, causing sedimentation upstream of a dam and “clean water” downstream (Assani and Petit 2004; Kondolf 1997). Clean water refers to water that has little to no suspended sediments.

Several studies have documented how dam-created reservoirs act as sediment sinks (Ahearn et al. 2005; Teodoru and Wehrli 2005). As fine particles settle out above dams, they can fill in cobble and boulder habitat and raise (aggrade) the stream bed (Bednarek 2001). Such structures can also increase the scouring of substrates. Kondolf (1997) describes clean water as sediment starved; as a result, there is the potential to scour and erode downstream environments as the stream tries to regain sediment equilibrium. Increased erosion and incision downstream can result in several impacts downstream of dams. Erosion can lower groundwater tables and affect riparian vegetation through reduced access to water (Gillilan and Brown 1997). Second, if erosion is extremely high, incision down to bedrock can occur and effectively reduce hyporheic and groundwater–surface water interactions (Assani and Petit 2004). Finally, increased erosion can cause bank failures, resulting in large sediment inputs and a loss of riparian vegetation (Dietrich et al. 1989; Kondolf 1997; Sear 1995). Impacts of altered groundwater–surface water interactions on fish and invertebrates are discussed in Section 7.1.2.1.5 (*Altered Groundwater–Surface Water Interactions*), and impacts from altered riparian vegetation are discussed in Section 7.1.4 (*Riparian Vegetation Modifications*).

The reduction in suspended sediment (and turbidity) directly downstream of a dam can also influence predation of those species waiting to pass over dam structures. Experiments have shown that white sturgeon larvae predation by prickly sculpin increased in the presence of low-turbidity water (Gadomski and Parsley 2005). This suggests that some species use sediment as cover to some extent. The effects of suspended sediments on fish and invertebrates are discussed in detail in Section 7.1.6.3 (*Altered Suspended Sediments and Turbidity*).

Impacts from altered sediment transport are not limited to the riverine environment; depending on the location of the dam and the river system, impacts on coastal ecosystems are also possible. The reduction of sediment supply to estuarine and coastal environments will change habitat quality and cause erosion of beaches that rely on sediment from rivers. For example, the lack of sediment supply from two large dams on the Elwha River, Washington, has contributed to a loss of beach and coastline habitat (DOI 1995).

Dam removal also alters sediment transport in a river. Because sediment is trapped upstream of a dam, removal of the dam will increase sediment downstream. Although the potential for increased suspended sediment downstream from dam removal is highly likely, the effects are often short term. The impact depends on the type of removal, time of the year, length of time the dam was present, flow rates, and flow velocities (Bednarek 2001). Studies have shown that sediment pulses from dam removal can migrate through a system in days to weeks to years (Bednarek 2001); in some cases, sediment releases are similar to a periodic storm event (Winter 1990). Dam removal is one possibility for restoring natural sediment transport in a riverine system, as discussed in more detail in Section 11 (*Habitat Protection, Conservation, Mitigation, and Management Strategies*).

7.1.2.1.4 Altered Substrate Composition

Coupled with alterations in sediment transport are changes to substrate composition. In this white paper, substrate composition is defined as those size proportions that comprise the substrate (e.g., fines, sand, gravel, cobble). Alterations in sediment transport can change aggradation and incision in a channel, as well as increase erosion (as described previously). These alterations in turn alter the composition of stream bed substrates. Because HCP species depend on the presence or absence of particular substrate types to support important life-history functions, changes in substrate composition can have direct and indirect effects on those species.

Increased velocities associated with dams can indirectly affect HCP species by causing local bed scour around structures and with a corresponding deposition of sediment downstream. Bed scour into a substrate of mixed particle sizes (e.g., sand and gravel) can selectively remove finer sediment and cause the substrate to coarsen. Likewise, deposition of the finer sediment downstream can bury organisms and cause the substrate to become finer.

Changes in substrate size downstream can affect spawning habitat. For example, salmon require a range of sediment sizes, and spawning success depends on how well they can move sediment to create a redd with their tail. As a result, different species use gravels of different size and can effectively move only certain size classes of sediment (Kondolf 1997; Kondolf and Wolman 1993; Kondolf, Sale, et al. 1993). Gravel and cobble substrate is optimal for spawning of white sturgeon because their sticky eggs may be covered in sandy substrate, thus reducing oxygen transport (Paragamian et al. 2001). Gravel substrate is also preferred spawning habitat for Dolly Varden (Kitano and Shimazaki 1995). Therefore, deposition of fines both upstream and downstream will affect HCP species that rely on gravel and cobble substrate (e.g., Dolly Varden, other salmonids, sturgeon) by covering spawning beds, reducing oxygen transport, and reducing the removal of metabolic wastes from the developing embryos.

Another potential substrate alteration associated with dams is the loss of large woody debris (LWD). Large woody debris upstream of a dam becomes trapped in the reservoir, reducing the downstream amount of this important substrate fraction. LWD provides numerous benefits in streams, as summarized in the Habitat Modifications white paper (Herrera 2007e). Large woody debris can control habitat complexity, and impacts from LWD losses are discussed in Section 7.1.3 (*Ecosystem Fragmentation*).

Finally, changing substrate can adversely affect the growth of aquatic vegetation. In riverine environments, rooted aquatic macrophytes are often associated with sandy substrates (Cushing and Allan 2001). In coastal and marine environments, eelgrass is incapable of growing in dominantly gravelly substrates (Koch 2001). As a result, changes in aquatic vegetation from substrate alterations can affect fish and invertebrate species by changing habitat and cover, thus affecting predation. For details regarding the impacts on fish and invertebrates as a result of aquatic vegetation disturbances, see Section 7.1.5 (*Aquatic Vegetation Modifications*).

7.1.2.1.5 Altered Groundwater–Surface Water Interactions

A high level of substrate fines in channel substrate from a dam may hinder the connection between surface and groundwater, limiting vertical and lateral connectivity between these two habitat types (Edwards 1998; Pusch et al. 1998). This lack of connectivity can degrade conditions for riparian zone vegetation, reducing LWD recruitment to the stream channel and subsequently limiting habitat-forming and maintaining processes. Effects on ecological functions and freshwater aquatic species associated with degraded connectivity between different riverine habitat elements are well documented (Bilby and Bisson 1998; Hershey and Lamberti 1992; Karr 1991; Kelsey and West 1998; Montgomery et al. 1999; Naiman et al. 1992; Reiman and McIntyre 1993; Stanford and Ward 1992; Stanford et al. 1996). Besides altering connections between surface and groundwater, changes in flow regime, sediment transport, and substrate composition all affect in-channel hyporheic exchange.

Hyporheic exchange, characterized by exchange between surface water and subsurface water in streams and rivers, is extremely important for the health of riverine systems (Jones et al. 1995; Mulholland et al. 1997; Sheibley, Duff, et al. 2003; Triska et al. 1989). Increased hyporheic exchange between surface and subsurface waters will benefit aquatic biota by increasing benthic dissolved oxygen levels and promoting solute uptake, filtration, and transformation. Studies have shown that the availability of dissolved oxygen to incubating salmonid embryos is dependent on hyporheic exchange (Geist 2000; Greig et al. 2007) and that the occlusion of this exchange through siltation can lead to hypoxia within redds and decreased embryo survival, as described in Section 7.1.2.1.4 (*Altered Substrate Composition*).

The hyporheic zone does more than promote oxygen exchange in subsurface sediments; it effectively acts as a filter and zone of biogeochemical transformations. Increased hyporheic exchange has been associated with nutrient uptake and transformation (Fernald et al. 2006; Lefebvre et al. 2005) and may attenuate the transport of dissolved and particulate metals (Gandy et al. 2007). Elevated metals and nutrients can both have negative ramifications for fish and invertebrate health. The impacts from increased nutrients are discussed in Section 7.1.6 (*Water Quality Modifications*).

The interface between flow within the hyporheic zone and the stream channel is an important buffer for stream temperature (Poole and Berman 2001a); therefore, the alteration of groundwater flow or hyporheic exchange can affect stream temperature. The magnitude of the influence depends on many factors, such as stream channel pattern and depth of the aquifer (Poole and Berman 2001a). Stream temperature has been shown to be an important factor in determining the suitability of habitats for aquatic species. For example, in Montana, the distribution and abundance of bull trout is influenced by hyporheic and groundwater–surface water exchange (Baxter and Hauer 2000). Consequently, female bull trout tend choose areas of groundwater discharge (i.e., cooler temperatures) for locating their spawning redds (Baxter and McPhail 1999). A detailed discussion of the impacts of temperature on fish and invertebrates is presented in Section 7.1.6.1 (*Altered Temperature Regime*).

7.1.2.2 Direct and Indirect Effects

Natural flow regime is important for many HCP species because they rely on certain flows for habitat access, foraging, and cover. As a result, altered flow variability will result in disconnected habitat, changes in species composition and diversity, and changes in the mixing of coastal estuaries. These changes will have direct and indirect effects on fish and invertebrates and are discussed in detail in Section 7.1.3 (*Ecosystem Fragmentation*).

In addition to natural flow variability, fish and invertebrates inhabiting riverine environments require certain flow velocities for spawning, rearing, and foraging. Increases in flow velocities could present potential barriers to fish migration or could exceed thresholds for certain life-history stages of some HCP species. Direct effects from altered velocities include stress to migrating species through increased activity, exhaustion, and delayed migration. Indirect effects include changes in habitat accessibility, habitat quality, and increased predation. For instance, leopard and Umatilla dace inhabit riverine environments where the velocities are less than 1.6 ft/sec (Wydoski and Whitney 2003). Exceeding this velocity as a result of water released from a dam would render the habitat unsuitable for these species.

Direct and indirect effects of altered flow velocities on invertebrates are not well understood and represent an area for further research. However, for the HCP invertebrate species that are filter feeders (e.g., California floater and western ridged mussel) or rely on stable substrate for habitat structure, altered sediment transport is likely more important than changes in flow velocities. If the altered flow velocity is causing changes in the sediment delivery to coastal environments, then estuarine species such as the Olympia oyster and northern abalone could also be affected. The impacts from altered sediment supply are discussed below.

Alteration of channel geometry has both direct and indirect effects on fish and invertebrates. Fish and invertebrates require certain widths and depths for habitat, spawning, and cover. For example, mountain suckers in Lost Creek, Utah, showed a preference for spawning depths of 4.3–11.8 inches (11–30 cm) (Wydoski and Wydoski 2002). In addition, Watters (1999) documented several studies where increased depths in impoundments upstream of dams led to a decline in mussel populations as a result of deeper water and altered water quality. Indirect impacts arising from the alteration of channel geometry include the modification of natural sediment transport, a reduction in habitat connectivity, and a reduction in habitat complexity. Impacts from changes in sediment transport are discussed below, and changes to habitat are discussed in Section 7.1.3 (*Ecosystem Fragmentation*).

Altered sediment transport will have direct and indirect effects on fish and invertebrate species. Erosion from clean water can cause bank failure and increased suspended sediments. For mussel species, siltation within the impoundment and scour downstream of the dam have been attributed to the decline of several native species in the United States (Watters 1999). The impacts from increased suspended sediments are discussed in Section 7.1.6.3 (*Altered Suspended Sediments and Turbidity*). Coastal environments can also be affected by the reduction in sediment supply. Shallow nearshore marine habitats (structured by tidal currents, wind, and input from terrestrial and freshwater sources) support at least one life-history stage of all of the marine and anadromous HCP species [i.e., all salmonid, rockfish species, cod, hake, Pacific herring, walleye pollock, Newcomb's littorine snail, and Olympia oyster (WDNR 2006a, 2006b)]. Therefore, the lack of sediment nourishment to coastlines represents a major impact mechanism for this subactivity type on spawning, rearing, and foraging of HCP species due to the loss of quality habitat.

Alteration of the substrate composition through coarsening or fining of the bed can have direct and indirect effects on HCP species. The ecological effects of substrate coarsening and fining on salmonids in riverine environments are well known. Far less is known about the effects of these disturbances on the life-history stages of other freshwater fish and invertebrate species. Large substrates, exceeding the maximum size mobilized by spawning salmonids, are avoided during redd building (Kondolf and Wolman 1993). Field observations have shown that salmonids can build redds where the average substrate size (D_{50}) is up to 10 percent of the average body length (Kondolf and Wolman 1993). The optimal range of spawning gravels for salmonids is listed in Table 7-2.

Table 7-2. Spawning gravel criteria for salmonids.

Gravel bed criteria	Small-bodied Salmonids <13.8 in (<35 cm)	Large-bodied Salmonids >13.8 in (>35 cm)
Dominant substrate particle size	0.3–2.5 in (8–64 mm)	0.6–5 in (16–128 mm)
Minimum gravel patch size	10.8 ft^2 (1 m^2)	21.5 ft^2 (2 m^2)

Adapted from (Schuett-Hames et al. 1996).
Note: Small-bodied salmonids include cutthroat trout; large-bodied salmonids include coho and Chinook salmon and steelhead trout.

Bed scour and substrate coarsening are often accompanied by an increase in the interlocking strength of bed particles and the threshold force necessary for bed mobility (Church et al. 1998; Konrad 2000; Lane 1955). Substrate fining can adversely affect salmonids and trout by clogging spawning gravels with fine sediment (Zimmermann and Lapointe 2005). Embryo mortality has been found to occur from poor water circulation and lack of oxygenation associated with the filling of intergravel pore spaces by fine sediment (Bennett et al. 2003; Chapman 1988; Cooper 1965; Lisle and Lewis 1992). In a study of spawning chum salmon in low-gradient, gravel-bed channels of Washington and Alaska, Montgomery et al. (1996) found that minor increases in the depth of scour caused by bed fining and a reduction in hydraulic roughness significantly reduced embryo survival.

The deposition of fine sediment can also adversely affect invertebrates (Wantzen 2006). Fine sediment particles may clog biological retention mechanisms such as the filtering nets of caddisfly larvae, or the filtering organs of mollusks. Additionally, overburden from increased deposition has been shown to adversely affect invertebrates having low motility (Hinchey et al. 2006). In a survey of native freshwater mussels in the United States and Canada, it was concluded that declines in populations were caused by habitat destruction, dams, siltation, and channel modifications (Williams et al. 1993). Furthermore, Watters (1999) summarized the effects of impoundments on mussel species in the United States, and deposition of silt within and downstream of impoundments has been linked to extinction of several mussel species nationwide. Further discussion of the effects of suspended sediments on HCP species is presented in Section 7.1.6.3 (*Altered Suspended Sediments and Turbidity*).

Finally, structures that alter groundwater dynamics and hyporheic exchange in riverine systems can directly affect fish and invertebrates in the short term by influencing water quality and habitat suitability or availability. In the long term, changes to groundwater exchange can generate indirect effects on fish and invertebrate species by affecting low-flow conditions (i.e., increasing the magnitude of periods of drought, resulting in reduced habitat availability and suitability, potential stranding, or desiccation), as well as by affecting water quality through warmer stream temperatures and decreased organic and nutrient inputs.

7.1.3 Ecosystem Fragmentation

Dam presence affects HCP species through direct fragmentation of ecosystems. Ecological connectivity is essential between riverine and riparian ecosystems (Stanford and Ward 1993). In riverine environments, connectivity is generally expressed in three dimensions: longitudinally (upstream–downstream), laterally (channel–floodplain), and vertically (channel–hyporheic zone [the interface between surface and groundwater]) (Stanford and Ward 1993). Impacts from dams on vertical connectivity are discussed in Section 7.1.2.1.5 (*Altered Groundwater-Surface Water Interactions*).

Upstream–downstream connectivity is the most obvious alteration, with a reduction in sediment and food resources, increased potential for predation, and altered temperatures. Because flow variations are dampened by dams, connectivity of the river with its floodplain is reduced. This can result in reduced nutrient cycling, loss of favorable habitat, increased predation, and altered temperatures. Altered hyporheic exchange has similar effects with changes in temperature, nutrient cycling, and loss of refugia for invertebrates. Furthermore, these alterations to ecosystems contribute to changes in species interactions and diversity.

7.1.3.1 Submechanisms of Impact

Ecosystem fragmentation is a broad category that includes (1) altered longitudinal connectivity; (2) altered river–floodplain connectivity; (3) altered groundwater–surface water interactions; (4) altered LWD transport and recruitment; and (5) altered community composition.

7.1.3.1.1 Altered Longitudinal Connectivity

Dams block the migration of aquatic species, which is one of their most detrimental impacts on riverine ecosystems. Longitudinal connectivity is vital to ecosystem health, as organisms rely on up–down transport in riverine systems in search of optimal conditions for feeding, cover from predators, spawning, and food resources.

It is well documented that fish populations suffer in the presence of dams through blocked migration routes. Studies have shown that Atlantic salmon (Chanseau et al. 1999) and shrimp (Greathouse et al. 2006) have difficulties migrating over dams and weirs. In the Fox River, Illinois, freshwater mussel distribution was limited upstream of a dam site (Tiemann et al. 2007). In the Pacific Northwest, salmon, sturgeon, bull trout, and lamprey populations have suffered declines from decreased migration in the presence of dams (Hicks et al. 1991; Jager 2006; Moser et al. 2002; Neraas and Spruell 2001). Cumulative effects of dams have been documented as well. In the Columbia River, only 3 percent of tagged Pacific lampreys were able to reach the upper part of the river after passing over three dams (Moser et al. 2002). With each dam passed, the number of successful individuals decreased. In addition, lamprey were observed attempting passage multiple times (Moser et al. 2002). More details on the cumulative effects of dams on fish and invertebrates are presented in Section 8 (*Cumulative Effects*).

Depending on the size of the dam, upstream migrations of fish can be completely blocked. In other cases, fish passage structures may allow some individuals to migrate upstream. Although fish passage facilities are constructed at some dams to allow longitudinal migration, delays in passage are common. These delays can result in increased risk of predation or poaching while fish wait for an opportunity to pass a given structure (Bednarek 2001). In addition, some fish undergo physiological changes if spawning migrations are delayed. For example, if they are delayed, American shad have been known to reabsorb their gonads when returning to the ocean, without releasing eggs and sperm (Dadswell 1996).

Anadromous salmonids, which begin to decline physically upon entry to fresh water, may experience decreased spawning fitness as a result of delayed passage, including potential mortality before spawning is completed (Caudill et al. 2007). Delayed migration of adult Chinook on the Columbia River has been documented at the Bonneville Dam since the 1950s (Schoning and Johnson 1956). Furthermore, juvenile migration downstream can be delayed as free-flowing reaches are transformed into slow-moving/slackwater reservoirs (Peven 1987). For a detailed analysis on the direct and indirect impacts on fish and invertebrates from fish passage facilities, see the Fish Passage white paper (Herrera 2007a).

Alteration of habitat complexity and structure can also fragment longitudinal connectivity. Upstream of a dam, habitat structure is altered from a free-flowing (lotic) habitat to a lake-like (lentic) habitat. In this case, lake-adapted organisms dominate and, in some cases, encourage salmon predation (Wik 1995). Downstream of a dam, habitat types (e.g., pools, riffles) can be altered as a result of the effects of changes in flow regime, sediment transport, and substrate composition on channel morphology (see Section 7.1.2 [*Hydraulic and Geomorphic Modifications*]). For example, after several decades of sediment trapping, the Warche River in Belgium showed significant losses of pool and riffle habitat downstream of a dam (Assani and Petit 2004). Habitat quality was also affected by the presence of dams on the Fox River (Illinois), and this led to reductions of several freshwater mussel species (Tiemann et al. 2007).

7.1.3.1.2 Altered River–Floodplain Connectivity

As discussed in Section 7.1.2.1.1 (*Altered Flow Regime*), dams reduce flow variability, which affects river–floodplain connectivity. Floodplains are habitat-rich areas with pools and side channels that are important nurseries for aquatic organisms (Bednarek 2001). Floodplains can act as nutrient sinks and carbon sources for adjacent channels (Tockner et al. 1999; Valett et al. 2005). In addition, oxbow lakes are important habitat for juvenile fish (Penczak et al. 2003). Consequently, floodplain–channel connection augments allochthonous carbon budgets in restored channels and engages habitat that would otherwise be inaccessible. Even low-occurrence flooding (e.g., a 2-year flood) is important to sustain floodplain habitat diversity and to support aquatic species (Thoms 2003). Besides a reduction in flow, the loss of LWD will contribute to river–floodplain connectivity losses. The direct and indirect effects from the loss of LWD are discussed in Section 7.1.3.1.4 (*Altered Large Woody Debris Transport and Recruitment*).

Floodplains are sites of high biodiversity that require periodic inundation from rivers. Dams (as well as diversions) reduce the occurrence of inundation flows and can alter floodplain ecology, contributing to mortality and poor health of aquatic organisms (Kingsford 2000).

7.1.3.1.3 Altered Groundwater–Surface Water Interactions

The impacts on fish and invertebrates from alteration of groundwater–surface water interactions are discussed in Section 7.1.2.1.5 (*Altered Groundwater–Surface Water Interactions*) under the hydraulic and geomorphic modifications impact mechanism. Overall, the impacts are identical, and this submechanism of impact is repeated here because the pathway of impact is different from Section 7.1.2.1.5. As discussed for hydraulic and geomorphic modifications, alterations in groundwater–surface water interactions are primarily the result of changes in sediment transport and substrate composition. For ecosystem fragmentation, the same impacts can result from a disconnection between the river and floodplain, as well as between the river and riparian zone.

7.1.3.1.4 Altered Large Woody Debris (LWD) Transport and Recruitment

Dams directly alter LWD dynamics by physically blocking the downstream transport of LWD. Indirectly, dams reduce LWD inputs through changes in river–floodplain connectivity. Woody debris in freshwater streams controls channel morphology, regulates the storage and transport of sediment and particulate organic matter, and creates and maintains hydraulic complexity that contributes to fish habitat (Murphy and Meehan 1991; Naiman et al. 2002). Within channels, approximately 70 percent of structural diversity is derived from root wads, trees, and limbs that fall into the stream as a result of bank undercutting, mass slope movement, normal tree mortality, or windthrow (Knutson and Naef 1997). In small streams, LWD is a major factor influencing pool formation in plane–bed and step–pool channels. Bilby (1984) and Sedell et al. (1985) found that approximately 80 percent of the pools in several small streams in southwest Washington and Idaho were associated with wood. In larger streams, the position of LWD strongly influences the size and location of pools (Naiman et al. 2002). In larger streams, LWD is typically oriented downstream due to powerful streamflow, which favors the formation of backwater pools along margins of the mainstem (Naiman et al. 2002). The hydraulic complexity created by LWD encourages the capture and sequestration of other allochthonous inputs, making these materials more available to the food chain through grazing and decomposition (Knutson and Naef 1997; Murphy and Meehan 1991; Naiman et al. 2002; Quinn 2005).

LWD accumulation creates habitat through two primary mechanisms. First, the wood itself provides cover and creates local scour pool habitat; second, wood in channels promotes lateral and vertical energy transfer, thus altering the composition and/or abundance of accessible food sources for HCP species, as well as creating more accessible habitat for foraging and rearing. The presence of LWD within channels creates locally complex geomorphic features, including scour pools and depositional bars. For example, studies have shown that channels with LWD retain more bedload (Faustini and Jones 2003) and particulate organic matter (POM) (Cordova et al. 2007; Diez et al. 2000) than similar reaches without wood. Depending on channel form and wood size and orientation, LWD-induced pools and bars can occur upstream, downstream, and/or lateral to wood structures (Abbe and Montgomery 2003; Gurnell and Petts 2006; Kail 2003).

LWD itself serves as a substrate for algal growth and macroinvertebrate habitat (Bowen et al. 1998). In German streams, Hoffman (2000) showed an intimate connection between LWD and all life-history stages of the lepidostomatid caddisfly, *Lasiocephala basalis*. Meanwhile, Hilderbrand et al. (1997) noted no change in macroinvertebrate populations following a wood addition experiment in Virginia, and Spanhoff et al. (2006) noted a net negative impact of wood addition on stream macroinvertebrates. Despite these findings, other studies have shown that wood in channels and on shorelines can serve as a substrate for both algal growth (Bowen et al. 1998) and macroinvertebrate habitat (Rolauffs et al. 2001; Warmke and Hering 2000).

The presence of LWD within channels has been shown to promote floodplain connection during storm flow conditions by increasing flow resistance within the channel (Dudley et al. 1998). Increased channel roughness promotes backwater conditions that locally connect the floodplain and channel habitat. Large woody debris also promotes floodplain connection by diverting flow into side channels (Abbe and Montgomery 1996). Overall, LWD increases access to floodplain habitat and promotes hyporheic exchange.

7.1.3.1.5 Altered Community Composition

Many of the submechanisms of impacts discussed so far (e.g., altered flow regime, altered connectivity, LWD transport and recruitment) will ultimately result in alterations to natural species composition and diversity. Species composition can be altered in the presence of dams as invasive species become more dominant than native species as a result of dramatic changes in the riverine ecosystem (Moyle 1976). The losses of native fauna as a result of dams can dramatically alter basal food resources and assemblages of invertebrate competitors and prey (Greathouse et al. 2006). In addition, changes in water quality, in particular temperature, can cause shifts from resident cool water species to warm water species. For mussel species, it has been shown that changes in temperature and increased siltation from dams have reduced the number of native species and increased the number of invasive species nationwide (Watters 1999).

In Japan, Katano et al. (2006) showed that the number of species, total diversity, and biomass of fish were lower above three small-scale dams (4.9–12.8 ft [1.5–3.9 m] in height) compared to downstream reaches. In addition, the authors found that food webs above dams were simpler relative to downstream communities. In New Zealand, species richness of different fish and crayfish were lower in areas upstream of dams (Joy and Death 2001). Food web structure and functional feeding groups of macroinvertebrates were changed above and below dams on the Grande River, Argentina (Vallania and Corigliano 2007). The authors found that collector-filterers, scrapers, and predators increased downstream, whereas the collector-gatherers and shredders decreased relative to upstream reaches. A similar result was observed in a series of mountain streams in Spain (Camargo et al. 2005).

Alterations to species composition and diversity from dams are not limited to freshwater environments. The lack of freshwater inputs from reduced flow regimes can affect species in estuarine and marine environments as well (Drinkwater and Frank 1994). On the Olympic Peninsula, the presence of dams on the Elwha River has reduced sediment transport to coastal environments, causing a shift to invasive kelp and barnacles in the receiving water body (DOI 1995). This reduction of sediment contributed to a loss of estuaries that serve as nurseries for fish and shrimp because sediment bars separating brackish water from the ocean are no longer present (DOI 1995).

7.1.3.2 Direct and Indirect Effects

Alterations in longitudinal connectivity will have direct impacts on fish as a result of blocking access to spawning habitat and delayed migration. Indirect effects include increased exposure to predation. In the case of adult salmonids, the cost of delayed migration and energy expenditure can have a demonstrable effect on survival, as well as spawning productivity. Caudill et al. (2007) examined the relationship between delayed migration, survival, and spawning productivity by using radiotelemetry to track the behavior and fate of Chinook salmon and steelhead navigating fish passage structures on Columbia River dams. Statistically correcting for other sources of mortality, they found a distinct inverse relationship between the time required for individual fish to transit fish ladders bypassing Columbia and Snake River dams, and survival to reach spawning grounds. While the drivers of this inverse relationship are complex, energy expenditure and stress associated with navigating the structures are primary contributing factors. However, extant natural barriers eliminated by impoundments are also energetically demanding (Brown et al. 2002). In combination with other stressors imposed by fish passage structures (e.g., prolonged exposure to elevated water temperatures, increased harvest, and predation pressure), the effects of migration delay induced by fish passage structures appear to be cumulatively significant (Caudill et al. 2007).

Little is known about the effects of longitudinal connectivity on invertebrate species. However, freshwater mussel larvae rely on attachment to host fish (Nedeau et al. 2005), so reduction in fish populations may have an indirect effect on mussels such as the California floater and western ridged mussel. In fact, several studies summarized by Watters (1999) showed that the loss of fish hosts has been linked to the decline of native mussel species.

Alteration of longitudinal connectivity and resulting changes to both upstream and downstream habitat complexity will impact HCP species because many of the HCP fish species require a range of habitat types throughout their life histories. In a survey of habitat use in Southeast Alaska, Dolly Varden were observed using step-pools (Bryant et al. 2007) and were more common in tributaries compared to the main channel (Bramblett et al. 2002). Steelhead trout were found to use main channel and side tributaries equally (Bramblett et al. 2002), and coho salmon were observed more often in tributaries (Bramblett et al. 2002). Radio-tagged Pacific lamprey in the John Day River (Oregon) were observed using the lateral margins of riffles and glides, and used boulders for cover (Robinson and Bayer 2005). In Montana, mountain suckers were shown to prefer riffles (Wydoski and Wydoski 2002). Loss of certain habitats may affect growth, survival, and fitness of many HCP species.

Floodplain connectivity creates fish forage and refuge habitat for several of the HCP species (Feyrer et al. 2006; Henning 2004). Chinook salmon that rear in floodplains have been shown to grow faster than those rearing in adjacent channels (Sommer et al. 2001). Green sturgeon tagged on the Rogue River (Oregon) showed use of off-channel coves (Erickson et al. 2002). Steelhead use both main channel and side channel habitat (Bramblett et al. 2002). Egg dispersal into newly inundated habitat has been hypothesized to increase the number of Columbia River white sturgeon (Coutant 2004). Therefore, the loss of floodplain habitat can lead to mortality, stranding, and decreased reproductive success for many of the HCP species.

Large woody debris (LWD) is important for instream habitat, and alteration of LWD will have direct and indirect effects on HCP species. Fish use complex environments and the structure of the LWD itself for cover and refuge (Cederholm et al. 1997; Everett and Ruiz 1993; Harvey et al. 1999). In a study of Smith Creek in northwest California, Harvey et al. (1999) found that tagged adult coastal cutthroat trout moved more frequently from pools without LWD than from pools with LWD. They hypothesized that the habitat created by LWD attracts fish, and once fish establish territory within the desirable habitat, they remain there longer. A study by Cederholm et al. (1997) on a tributary of the Chehalis River (Washington) found that LWD additions caused an increase in winter populations of juvenile coho salmon and age-0 steelhead populations. It should be noted that Fausch et al. (1995) and others have criticized studies such as Harvey et al. (1999) because it is difficult to determine if increased abundance in treatment sites is due to increased populations or simply just concentrations of fishes that would have thrived equally well in other habitat. Nonetheless, studies have documented fish species utilizing LWD for critical habitat, such as Dolly Varden, coho, steelhead, and cutthroat trout (Bryant et al. 2007).

An indirect impact from the loss of LWD is an increase in nutrient loading to downstream receiving waters. Channel complexity promotes the retention of water and organic material. This retention plays an important role in the fate of nutrients in the stream channel. In a study by Mulholland et al. (1985), it was suggested that leaf litter in streams promotes nutrient retention as the leaf pack acts as a substrate for nutrient-hungry microbes. Using solute injection techniques, Valett et al. (2002) found that phosphorus uptake in channels with high LWD volumes, frequent debris dams, and fine-grained sediments was significantly greater than in channels in younger forests without these characteristics. Corroborating this finding, Ensign and Doyle (2005) conducted phosphorus injections in streams both before and after the removal of LWD and coarse-particulate organic matter (CPOM) in the channels and found that phosphate uptake decreased by up to 88 percent after LWD removal. These studies show that channel complexity increases water retention and, through CPOM and LWD retention, provides a substrate for biofilm growth. Decreased nutrient retention affects both local waterways and downstream receiving waters. Local waterways are affected through the associated reduction in primary production, and receiving waters (which are primarily located in more nutrient-impacted lowland areas) are affected through additional nutrient loading, which may lead to eutrophication. Impacts on fish and invertebrates resulting from eutrophication are discussed in Section 7.1.6.6 (*Altered Nutrient Loading*).

Direct and indirect effects from alteration in species diversity include a loss of food resources from changes in macroinvertebrate assemblages and increased predation caused by shifts to nonnative species. In addition, changes to water quality can result in alterations to species interactions. For example, suspended sediment can be used as cover, and predation of white sturgeon larvae by prickly sculpin has been shown to increase in the presence of low-turbidity water (Gadomski and Parsley 2005).

7.1.4 Riparian Vegetation Modifications

Dams alter riparian vegetation through several pathways. First, construction and maintenance activities may directly remove or disturb riparian vegetation. Second, changes in sediment transport may result in bank failure and loss of riparian vegetation through increased erosion. Removal or disturbance of riparian vegetation during construction activities permitted under HPAs can expose HCP species to stressors caused by a variety of submechanisms of impact and are discussed below.

7.1.4.1 Submechanisms of Impact

Removal or modification of riparian vegetation will result in the following submechanisms of impact:

- Altered shading, solar input, and ambient air temperature
- Altered bank and shoreline stability
- Altered allochthonous inputs
- Altered groundwater–surface water interactions
- Altered habitat complexity
- Increased nutrient/pollutant loading.

These impact mechanisms and related ecological stressors are described below.

7.1.4.1.1 Altered Shading, Solar Input, and Ambient Air Temperature

Flow control structure projects can influence riparian vegetation and shading through riparian vegetation loss during construction and maintenance activities. During construction, vegetation will be removed for equipment access and construction of dam facilities. In addition, the dam itself will indirectly lead to modifications in riparian vegetation through alterations in sediment transport, which can cause bank and shoreline instability (see the next section for details on this submechanism of impact).

Removal of riparian vegetation as part of flow control structure projects affects water temperature in riverine environments through a number of mechanisms. The dominant effect pathway is that of reduced shading on solar radiation exposure. The influence of shade on water temperature generally diminishes as the size of the stream increases because of the proportionally reduced area in which riparian vegetation can insulate against solar radiation and trap air next to the water surface (Knutson and Naef 1997; Murphy and Meehan 1991; Poole and Berman 2001a; Quinn 2005). Alternatively, riparian vegetation removal and alteration can cause surface waters to gain or lose heat more rapidly because the ability to regulate ambient temperatures is reduced (Bolton and Shellberg 2001; Knutson and Naef 1997; Murphy and Meehan 1991; Poole and Berman 2001a; Quinn 2005).

In addition to the effects of shading, a broad array of research indicates that alterations of riparian vegetation can strongly affect temperatures even when adequate stream shading is still provided. Riparian vegetation restricts air movement, providing an insulating effect that regulates ambient air temperatures. Alterations of the riparian buffer width and vegetation composition can degrade this insulating effect, leading to greater variability in ambient air temperatures that in turn influence water temperatures (AFS and SER 2000; Bartholow 2002; Barton et al. 1985; Beschta 1991, 1997; Beschta et al. 1988; Beschta and Taylor 1988; Brosofske et al. 1997; Brown 1970; Chen et al. 1992, 1993, 1995; Chen et al. 1999; Johnson and Jones 2000; MacDonald et al. 2003; May 2003; Murphy and Meehan 1991; Spence et al. 1996; Sridhar et al. 2004; Sullivan et al. 1990; Theurer et al. 1984; USFS et al. 1993). For example, Chen et al. (1995) found that maximum air temperatures at the margins of old-growth forest stands are elevated 3–29°F (2–16°C) relative to interior temperatures. Riparian buffer widths of 100–300 ft may be necessary to provide full ambient temperature regulation (AFS and SER 2000; Brosofske et al. 1997). Additional information on appropriate buffer widths is provided in Section 11.1.2 (*Riparian Vegetation Modifications*) under Section 11 (*Habitat Protection, Conservation, Mitigation, and Management Strategies*).

Alteration of the temperature regime in riverine and lacustrine systems due to the alteration of riparian vegetation is a well-documented stressor on native fish populations. For example, in a study of 12 streams in Japan, it was shown that forest practices (i.e., logging) resulted in increased temperatures and decreased abundance of resident Dolly Varden (Kishi et al. 2004). Loss or degradation of the shading and ambient temperature regulation functions provided by riparian vegetation can increase water temperatures in summer when solar radiation exposure and ambient air temperatures are highest. In winter, loss or degradation of the insulating capacity of riparian vegetation can decrease water temperatures and increase the incidence of ice scour. Increased stream temperatures can also cause a concomitant decrease in dissolved oxygen levels, an additional stressor with additive deleterious effects. Numerous studies of such effects in the field have documented the deleterious effects on fish species of changes in the temperature regime in freshwater stream systems. However, some studies showed that in light-limited streams, selective thinning of forests can have a positive effect on fish. In northern California, cutthroat and rainbow trout responded positively to increased light from riparian thinning through increased primary productivity that stimulated the food web (Wilzbach et al. 2005). A detailed discussion of the impacts of altered dissolved oxygen concentrations and altered temperatures on fish and invertebrates is presented in Section 7.1.6 (*Water Quality Modifications*).

7.1.4.1.2 Altered Bank and Shoreline Stability

Many HPA-permitted activities involve the temporary or permanent modification of riparian vegetation structure. Riparian vegetation is an important component of the aquatic ecosystem that serves a variety of functions for habitat structure, water quality, and biological productivity.

The specific nature of these functions varies depending on the type of environment, but increasing bank cohesion plays an important role in regulating channel width and substrate.

The root structure supporting riparian vegetation naturally resists the shear stresses created by flowing water and thus retards bank erosion, stabilizing stream banks and shorelines, and maintaining valuable habitat features along stream margins, such as undercut banks. By dissipating the erosive energy of flood waters, wind, and rain, and by filtering sheet flows, riparian vegetation limits the amount of fine sediment entering river and stream systems (Brennan and Culverwell 2004; Knutson and Naef 1997; Levings and Jamieson 2001). If riparian vegetation is removed as part of an HPA-permitted activity, stream banks and shorelines will likely be exposed to the erosive effects of wind, rain, and current. The removal of riparian trees and understory can dramatically alter stream bank stability and the filtering of sediments from overland flow (Kondolf and Curry 1986; Shields 1991; Shields and Gray 1992; Simon 1994; Simon and Hupp 1992; Waters 1995), increasing erosion and inputs of fine sediment (Bolton and Shellberg 2001).

7.1.4.1.3 Altered Allochthonous Input

Riparian detritus and other externally derived (allochthonous) materials are the primary sources of organic matter in headwater streams, forming the basis of the food web (MacBroom 1998). This material includes terrestrial macroinvertebrates along with leaves, branches, and other vegetative materials, the latter providing food sources for benthic macroinvertebrates (Bilby and Bisson 1998; Knutson and Naef 1997; Murphy and Meehan 1991). As rivers increase in order and grow in size, these materials are processed and recycled by an increasing diversity of organisms (Vannote et al. 1980). Without allochthonous inputs, the forage detritus available for benthic macroinvertebrates is compromised, also diminishing the habitat and species diversity of these prey items (Murphy and Meehan 1991). Removal of freshwater riparian vegetation as part of HPA-permitted activities would cause an incremental decrease in the input of allochthonous materials to the nearby aquatic environment and food web.

7.1.4.1.4 Alteration of Groundwater–Surface Water Interactions

Alteration or removal of riparian vegetation would appreciably change the interface between plants, soil, and water on and near the bank surface. Riparian vegetation acts as a filter for groundwater, removing sediments and taking up nutrients (Knutson and Naef 1997). In conjunction with upland vegetation, riparian vegetation moderates streamflow by intercepting rainfall, contributing to water infiltration, and using water via evapotranspiration. Plant roots increase soil porosity and vegetation helps to trap water flowing on the surface, thereby aiding in infiltration as the water stored in the soil is later released to streams through subsurface flows. Through these processes, riparian and upland vegetation help to moderate storm-related flows and reduce the magnitude of peak flows and the frequency of flooding. Riparian vegetation, the litter layer, and silty soils absorb and store water during wet periods and release it slowly over a period of months, maintaining streamflows during low rainfall periods (Knutson and Naef 1997).

HPA-permitted activities that create a physical barrier between the bank and hyporheic flow (e.g., riparian vegetation removal) may prevent exchange between the bank and with the aquatic ecosystem. Because the interface between flow within the hyporheic zone and the stream channel is an important buffer for stream temperatures (Poole and Berman 2001b), alteration of groundwater flow can affect stream temperature. The magnitude of the influence depends on many factors, such as stream channel pattern and depth of the aquifer (Poole and Berman 2001b).

Indirect impacts on HCP species from the reduction of groundwater–surface water interactions include increased nutrient loading, altered temperatures, and habitat changes. Any activity that affects riparian or nearshore areas will degrade the buffering capability of the terrestrial–aquatic ecotone. Numerous studies have shown that wide stream buffers are effective at attenuating nutrients (Feller 2005; Mayer et al. 2005), herbicides (Gay et al. 2006), and sediment loading (Jackson et al. 2001). Riparian vegetation retards overland flow, promotes infiltration, and assimilates shallow groundwater nutrients. When this vegetation is removed through any HPA-permitted activity, nutrients and pollutants will be more efficiently transported from upland sources to downgradient water bodies. Forested buffers can effectively remove nutrients in shallow groundwater. In a study of a forested buffer in Alabama, a 33-ft (10-m) buffer reduced the groundwater nitrate concentration by 61 percent (Schoonover and Williard 2003). In a subsequent study of a forested wetland buffer, a buffer averaging 125 ft (38 m) wide reduced the nitrate concentration by 78 percent and total phosphorus by 66 percent (Vellidis et al. 2003).

7.1.4.1.5 Altered Habitat Complexity

Some effect on habitat complexity inevitably occurs when riparian vegetation is removed as part of flow control projects (in both freshwater and marine habitats). Because LWD is derived from riparian vegetation inputs, reduction in riparian vegetation will alter LWD dynamics of the system. For example, logging practices in northwestern Montana have been shown to decrease habitat complexity through the reduction of LWD inputs (Hauer et al. 1999). In particular, the authors observed a reduction in pools formed by scour around wood. The importance of LWD and a detailed discussion of the impacts of LWD reduction on fish and invertebrates are provided in Section 7.1.3.1.4 (*Altered Large Woody Debris Transport and Recruitment*).

Riparian vegetation is also important for bank-side habitat and cover from predation, as well as temperature. For example, the use of submerged riparian vegetation during early development has been hypothesized to increase Columbia River white sturgeon recruitment (Coutant 2004). In another study, radio-tagged cutthroat trout were observed using pools associated with LWD for cover (Harvey et al. 1999). In addition, undercut banks are stabilized by riparian vegetation roots, which reduce the effects of erosion from streamflow. These undercut banks provide shade and lower temperatures, which are important for fish habitat and cover from predation (Angermeier and Karr 1984; Heggenes and Borgstrom 1988; Rowe et al. 2004). Therefore, significant reduction in riparian vegetation will affect species composition.

7.1.4.1.6 Altered Nutrient/Pollutant Loading

Dams can alter riparian vegetation through direct removal during construction and maintenance activities, or indirectly through increased erosion and resulting bank failure. As stated earlier, riparian vegetation filters groundwater by removing sediments from overland flow and taking up nutrients in groundwater flow (Knutson and Naef 1997). Loss of riparian vegetation will result in a decline in the ecosystem's ability to process nutrients and increase nutrient loading to surface water. In addition, loss of riparian vegetation will change the natural hyporheic exchange through changes in groundwater discharge to rivers. Through these pathways, increased nutrient loading may result and can lead to problems of eutrophication and lowered dissolved oxygen concentrations in receiving waters. The direct and indirect effects on HCP species from this process are discussed in Sections 7.1.6.6 (*Altered Nutrient Loading*) and 7.1.6.2 (*Altered Dissolved Oxygen*).

7.1.4.2 Direct and Indirect Effects

The direct effects of a loss of riparian vegetation are twofold. First, modification of riparian vegetation will have an effect on water temperatures and levels of ambient solar input. Temperature is a critical factor for many of the HCP species, which require a certain thermal regime for survival. Extreme temperatures can lead to mortality, reduced foraging, and decreased spawning success. The direct and indirect effects from altered temperature regime are discussed in Section 7.1.6 (*Water Quality Modifications*). Second, slope instability can increase turbidity by delivering excess fine-grained sediment to a river, affecting water quality and habitat conditions. The effects associated with turbidity are discussed in Section 7.1.6 (*Water Quality Modifications*). In addition, slumping of unstable banks caused by a loss of riparian vegetation can bury invertebrates and larvae. Although some specifics are known for marine invertebrate species (Hinchey et al. 2006), it is unknown what tolerance limits the HCP freshwater mollusks may have with respect to burial.

Riparian vegetation is a known source of organic matter, nutrients, and macroinvertebrate prey items for HCP species, and the recruitment of these materials is diminished when riparian vegetation is removed or modified (Brennan et al. 2004; Lemieux 2004; Maser and Sedell 1994; Miller et al. 2001; Sobocinski 2003; Williams et al. 2001). Sobocinski (2003) has documented the importance of insect communities and benthic infauna that are either a direct or indirect result of riparian vegetation. These lower trophic organisms serve as the basis of the food web, and a reduction in allochthonous food sources to rivers diminishes the ability of the system to support higher trophic organisms, including most of the HCP fish species that use the riverine environment. As in riverine environments, a reduction of allochthonous food sources to marine and lacustrine environments diminishes the ability of these systems to support higher trophic organisms. This would extend to most of the marine and estuarine HCP species.

The effects of altering groundwater–surface water interactions are discussed in depth in Section 7.1.2.1.5 (*Alteration of Groundwater–Surface Water Interactions*). Impacts from increased nutrient loading stem from a reduction in dissolved oxygen concentrations and are discussed in detail in Section 7.1.6.6 (*Altered Nutrient Loading*). Impacts from altered temperature include delayed migration, reduction of suitable habitat, developmental problems, and mortality, as discussed in detail in Section 7.1.6.1 (*Altered Temperature Regime*).

The loss of habitat complexity from LWD will result in direct and indirect impacts on fish and invertebrate species. Riparian vegetation loss will reduce LWD, and impacts from this process are discussed fully in Section 7.1.3.1.4 (*Altered Large Woody Debris Transport and Recruitment*). Loss of habitat complexity from undercut bank loss will alter temperatures, cover, and potentially increase predation, leading to mortality. The effects of increased temperature are discussed in Section 7.1.6.1 (*Altered Temperature Regime*).

7.1.5 Aquatic Vegetation Modifications

Loss of aquatic vegetation will result in reduced autochthonous (in-stream) production, which provides important energy sources in aquatic food webs. In addition, aquatic vegetation loss can reduce habitat complexity through a reduction in cover for fish species, as well as changes in surface water flow patterns. Finally, because aquatic vegetation uses nutrients for growth, a reduction in aquatic vegetation will alter nutrient loading within stream and river ecosystems. Dams can cause losses of aquatic vegetation by several pathways. Increased velocities can scour algae downstream and damage macrophytes, reducing cover for fish. Second, changes in substrate composition with an increase in fine sediment transport can bury aquatic vegetation. Finally, modification may occur directly from construction and maintenance activities. These modifications in aquatic vegetation will lead to impacts on HCP species.

7.1.5.1 Submechanisms of Impact

Removal or modification of aquatic vegetation will result in the following submechanisms of impact:

- Altered autochthonous production
- Altered habitat complexity
- Altered nutrient/pollutant loading.

These impact mechanisms and related ecological stressors are described below.

7.1.5.1.1 Altered Autochthonous Production

Aquatic primary producers, such as benthic algae, macrophytes, and phytoplankton, play key roles in the trophic support of stream ecosystems. In general, benthic algae occur in the form of microscopic unicellular algae, forming thin layers or assemblages called periphyton. Macrophytes include angiosperms rooted in the stream bottom, along with mosses and other bryophytes. These include many forms such as rooted plants with aerial leaves, floating attached plants with submerged roots, floating unattached plants, and rooted submerged plants (Murphy 1998). A small algal biomass in a stream can support a much larger biomass of consumers due to the rapid turnover in biomass (Hershey and Lamberti 1992; Murphy 1998). Although aquatic primary production is sometimes underrated due to the small amount of algae and plants present in many streams, it is a basic energy source for freshwater ecosystems.

The uptake of carbon, nitrogen, and phosphorous by aquatic vegetation provides important nutrients to fish and invertebrate consumers. This aquatic primary production is the source of autochthonous (instream) organic matter and part of the source of allochthonous (terrestrial) matter in each stream reach. Invertebrate grazing of these primary producers by snails, caddisflies, isopods, minnows, and other organisms is an important pathway of energy flow. For stream herbivores, for example, benthic diatoms are the most nutritious and easily assimilated food source (Lamberti et al. 1989). The availability of algae regulates the distribution, abundance, and growth of invertebrate scrapers (Hawkins and Sedell 1981), an important food source for fish. As drift-feeders, juvenile salmonids focus on food from autochthonous pathways. Invertebrate scrapers and collector–gatherers are known to be most frequently eaten by salmonids (Bilby and Bisson 1992; Hawkins et al. 1983; Murphy and Meehan 1991). Although terrestrial and adult aquatic insects are important (Bjornn and Reiser 1991), juvenile salmon in streams have been found to be primarily supported by autochthonous organic matter (Bilby and Bisson 1992).

7.1.5.1.2 Altered Habitat Complexity

Freshwater macrophytes are also known to contribute to habitat complexity by changing surface water patterns, slowing water flow, trapping sediments, and altering temperature and water chemistry profiles. Through the trapping of particles by plant fronds, they also change the nature of the surrounding sediments by increasing the organic matter content and capturing smaller grain size sediment than normally occurs in uncolonized areas (Carrasquero 2001). In addition, submerged aquatic vegetation has been shown to increase hyporheic exchange (White 1990), which in turn will promote nutrient cycling.

7.1.5.1.3 Altered Nutrient/Pollutant Loading

Numerous studies have shown that macrophytes and algae in both marine and freshwater environments reduce ambient concentrations of suspended sediment (Abdelrhman 2003; Moore 2004), nutrients (Moore 2004), and metals (Fritioff and Greger 2003). In a study of macrophyte effects on sediment and nutrient retention in Danish streams, Sand-Jensen (1998) reported that dense-stemmed macrophytes created conditions conducive to sediment deposition and that the sediments retained within the macrophyte stands were fine-grained and nutrient-rich. He noted that enrichment of sediment within macrophyte beds relative to the surrounding substratum was 0.1597 lb organic matter per ft2 (780 g /m2), 0.006 lb nitrogen per ft2 (30 g/m2), and 0.005 lb phosphorus per ft2 (25 g per m2). Therefore, any modification of aquatic vegetation from the presence of a dam will likely result in increased suspended sediments, increased nutrient loading, and changes in hyporheic exchange, all adversely affecting HCP species.

7.1.5.2 Direct and Indirect Effects

In marine environments, seagrasses have been linked to improved water quality. As an example, Moore (2004) noted decreased nutrient concentrations and turbidity levels in seagrass beds relative to areas outside the beds along the littoral zone of the Chesapeake Bay National Estuarine Research Reserve. But aquatic vegetation not only reduces nutrient and sediment concentrations; the plants themselves can sequester harmful trace metal pollutants and are frequently planted in wetland treatment systems with that intended function. In a comparative study of heavy metal uptake in terrestrial, emergent, and submerged vegetation, Fritioff and Greger (2003) noted that submerged vegetation was efficient at removing zinc, copper, cadmium, and lead from influent stormwater.

As a result of the many benefits of aquatic vegetation described above, the loss of aquatic vegetation in riverine environments poses both direct and indirect effects on HCP species. Many of these species depend on aquatic vegetation for any one of their life-history stages, such as green and white sturgeon, California floater and western ridged mussel, mountain sucker, giant Columbia River limpet, pygmy whitefish, leopard and Umatilla dace, bull trout, and Pacific salmon (Frest and Johannes 1995; Hughes and Peden 1989; Mongillo and Hallock 1998; Mongillo and Hallock 1999; Watters 1999). More specifically, adhesive eggs of the Olympic mudminnow rely on attachment to aquatic vegetation for egg and larval development (Coutant 2004).

The density of coho salmon fry in the summer has been found to be directly related to the abundance of algae. A high density of fry can result from smaller feeding territories (Dill et al. 1981) due to increased invertebrate prey (Hawkins et al. 1983; Murphy et al. 1981). Increases in vertebrate production have been found to occur primarily in the spring and early summer, coincident with the primary production cycle of benthic algae (Murphy 1998). Therefore, the removal of or permanent disturbance to algal communities could have an adverse effect on local freshwater ecosystems and the HCP species that depend on these ecosystems. For coho salmon fry, the reduction in prey area (i.e., smaller feeding territories) results in a direct effect on fitness, growth, and survival.

The direct and indirect effects of aquatic vegetation removal on invertebrates is less well known. However, the California floater in the Eel River (California) is commonly associated with aquatic vegetation, which is used for protection from high flows (Howard and Cuffey 2003). This would imply that the loss of aquatic vegetation for some invertebrates would result in impacts on those species using this habitat.

Dam removals have shown contrasting results with respect to impacts on aquatic vegetation. In one study, the removal of a dam resulted in increases in aquatic macrophytes leading to increased cover and habitat for fish (Hill et al. 1993). However, dam removal may kill off some vegetation from sediment released during the removal process and subsequently cause abrasion of roots and stems (Wood and Armitage 1997). Dam removals may increase the scour of algae and insects, thereby altering food web interactions and food quality, particularly if algae or leaf accumulations are buried (Doeg and Koehn 1994; Newcombe and MacDonald 1991; Wood and Armitage 1997).

Finally, any activity that mechanically removes or by other means affects aquatic vegetation will reduce the sediment, nutrient, and pollutant retention and reduction capabilities of the system. Indirect impacts from the removal of aquatic vegetation will cause increased nutrient and pollutant loading to receiving waters, which could exacerbate eutrophic conditions and/or metals toxicity. A detailed discussion of the impact on various species from nutrient loading is presented in Section 7.1.6.6 (*Altered Nutrient Loading)*.

7.1.6 Water Quality Modifications

The construction and maintenance of dams and their hydraulic and geomorphic impacts can give rise to a number of water quality modifications. In general, these modifications alter the following primary water quality variables: temperature, dissolved oxygen, suspended sediments (turbidity) and contaminated sediments, pH levels, nutrient and pollutant loading, introduction of toxic substances, and altered salinity. These water quality modifications represent many of the direct stressors for the HCP species; although the path to these stressors may vary, they are common among all subactivity types addressed in this white paper. As a result, the direct and indirect effects on HCP species are presented in this section in a broad sense to cover all types of flow control structures that are referred to often throughout the text. Where additional subactivity-specific water quality modifications are important, they are addressed in the appropriate *Water Quality Modifications* section for the specific subactivity type.

7.1.6.1 Altered Temperature Regime

Dams alter the thermal regime of a river through the release of water from the upstream reservoir. Because water above the dam is relatively stagnant compared to the flowing reach downstream, water in an impoundment will typically absorb heat and become stratified. Depending on where the water is released from the reservoir, it will either increase (upper water column reservoir releases) or decrease (lower water column reservoir releases) stream temperatures downstream. Water temperature is also strongly dependent on mixing in rivers and streams (Fischer et al. 1979). Stratification within rivers can reduce both habitat complexity and connectivity; stratified waters can lead to elevated surface temperatures, particularly during the summer months (Fischer et al. 1979). Temperatures have been shown to regulate nutrient cycling processes in streams (Sheibley, Duff, et al. 2003; Sheibley, Jackman, et al. 2003). In these studies, the authors showed through modeling, field monitoring, and laboratory experiments that coupled nitrification-denitrificaiton reactions were controlled by stream temperature (Sheibley, Duff, et al. 2003; Sheibley, Jackman, et al. 2003). In winter, nitrification-denitirfication reactions were suppressed and more nitrogen from groundwater discharge entered the stream channel. In summer, nitrification-denitrification reactions were more efficient, and very little nitrogen from groundwater discharge was observed in the surface water. Therefore, temperature alterations may also affect the nutrient concentration in rivers. Finally, an altered temperature regime can shift species composition from cool water to warm water species (Bednarek 2001).

7.1.6.1.1 Direct and Indirect Effects

Temperature is a primary metric of aquatic ecosystem health, and aquatic organisms have adapted to live within specific thermal regimes. Alterations to these thermal regimes occur at the detriment of local organisms. Thermal stress can occur through multiple direct and indirect pathways in fish and invertebrates. These include direct mortality, altered migration and distribution, increased susceptibility to disease and toxicity, and altered development, spawning, and swimming speeds (Sullivan et al. 2000). Motile organisms have the ability to avoid or evacuate those areas of extreme temperature, but even then the stress induced from periodic exposure and resulting habitat avoidance can affect organism health and contribute to mortality (Groberg et al. 1978). Each of the HCP species is ectothermic (cold-blooded); consequently, temperature is a resource that organisms use for energetic means. With organism metabolism dependent on water temperature, thermal regime may be the single-most important habitat feature controlling aquatic organisms.

Most research on temperature impacts on aquatic species has focused on salmonids. Different species of salmonids have evolved to use different thermal regimes. Despite these differences, the majority of salmonids prefer the same temperature ranges during most of their life-history stages. The primary exception to this is that char (bull trout and Dolly Varden) require lower temperatures for optimal incubation, growth, and spawning (Richter and Kolmes 2005). An optimal temperature matrix is presented in Table 7-3 (although different species have different requirements at various life-history stages).

These same temperature ranges have been adopted by the Washington Department of Ecology (Ecology) and are incorporated into the state water quality standards (WAC 173-201A 2006). Table 7-4 presents the highest 7-day average maximum thresholds as promulgated in the state standards.

Elevated water temperatures can also impair adult migration and spawning. Adult migration blockages occur consistently when temperatures exceed 70–72^{o}F (21–22^{o}C) (Poole and Berman 2001a). Thermal barriers to migration can isolate extensive areas of potentially suitable spawning habitat and contribute to prespawning mortality. If salmon are exposed to temperatures above 57^{o}F (14^{o}C) during spawning, gametes can be severely affected, resulting in reduced fertilization rates and embryo survival (Flett et al. 1996). Ideal temperatures for salmonid spawning are in the range of 44–57^{o}F (7–14^{o}C) (Brannon et al. 2004; McCullough et al. 2001).

Table 7-3 indicates that water quality thresholds for some life-history stages are considerably lower than the lethal limit. Fish are susceptible to a number of sublethal effects related to temperature. For instance, elevated but sublethal temperatures during smolting may result in desmoltification, altered emigration timing, and emigration barriers. Temperatures that impair smolting are above a range of between 52 and 59^{o}F (11 and 15^{o}C) (Poole and Berman 2001a; Wedemeyer et al. 1980). Temperatures in this range have been shown to reduce the activity of gill ATPase (McCullough et al. 2001), an enzyme that prepares juvenile fish for osmoregulation in saline waters (Beeman et al. 1994). Temperature-induced decreased gill ATPase has been correlated with a loss of migratory behavior in numerous salmonid species (Babanin 2006;

Marine and Cech 2004; McCormick et al. 1999) and constitutes a significant impairment to juvenile survival.

Table 7-3. Estimates of thermal conditions known to support various life-history stages and biological functions of bull trout (a species extremely intolerant of warm water) and anadromous (ocean-reared) salmon.

Consideration	Anadromous Salmon	Bull Trout
Temperature of common summer habitat use	10–17°C (50–63°F)	6–12°C (43–54°F)
Lethal temperatures (1-week exposure)	Adults: >21–22°C (70–72°F)	—
	Juveniles: >23–24°C (73–75°F)	Juveniles: 22–23°C (72–73°F)
Adult migration	Blocked: >21–22°C (70–72°F)	Cued:10–13°C 50–55°F)
Swimming speed	Reduced: >20°C (68°F)	—
	Optimal: 15–19°C (59–66°F)	—
Gamete viability during holding	Reduced: >13–16°C (55–61°F)	—
Disease rates	Severe: >18–20°C (64–68°F)	—
	Elevated: 14–17°C (57–63°F)	—
	Minimized: <12–13°C (54–55°F)	—
Spawning	Initiated: 7–14°C (45–57°F)	Initiated:<9°C (48°F)
Egg incubation	Optimal: 6–10°C (43–50°F)	Optimal: 2-6°C (36–43°F)
Optimal growth	Unlimited food: 13–19°C (55–66°F)	Unlimited food: 12–16°C (54–61°F)
	Limited food: 10–16°C (50–61°F)	Limited food: 8–12°C (46–54°F)
Smoltification	Suppressed: >11–15°C (52–59°F)	—

Source: (Poole et al. 2001).

Note: These numbers do not represent rigid thresholds, but rather represent temperatures above which adverse effects are more likely to occur. In the interest of simplicity, important differences between various species of anadromous salmon are not reflected in this table, and requirements for other salmonids are not listed. Likewise, important differences in how temperatures are expressed are not included (e.g., instantaneous maximums, daily averages).

Table 7-4. Aquatic life temperature criteria in fresh water.

Category	Highest 7-DADMax[a]
Char spawning	9°C (48.2°F)
Char spawning and rearing	12°C (53.6°F)
Salmon and trout spawning habitat	13°C (55.4°F)
Core summer salmonid habitat	16°C (60.8°F)
Salmonid spawning, rearing, and migration	17.5°C (63.5°F)
Salmonid rearing and migration **Only**	17.5°C (63.5°F)
Nonanadromous interior redband trout	18°C (64.4°F)
Indigenous warm water species	20°C (68°F)

Source: WAC 173-201A 2006, Table 200(1)(c).

[a] Except where noted, water temperature is measured by the 7-day average of the daily maximum temperatures (7-DADMax). Table 200(1)(c) lists the temperature criteria for each of the aquatic life use categories.

A substantial amount of information is available regarding tolerances of HCP species (particularly salmonids) to thermal stress. For instance, it has been found that coho egg, alevin, and fry development is most rapid at 39°F (4°C), while alevin and fry of pink and chum salmon develop fastest at 46°F (8°C) (Beacham and Murray 1990). Additional studies, mainly in the laboratory, have developed limits for other HCP species. Wagner et al. (1997), showed that rainbow trout mortality occurred at temperatures of 67.8–73.0°F (19.9–22.8°C). Temperatures above 71.6°F (22°C) can cause deformities in developing white sturgeon, while the best development performance occurs between 59 and 66.2°F (15 and 19°C) (Mayfield and Cech 2004). Furthermore, elevated temperatures can make white sturgeon more susceptible to infection from viruses (Watson et al. 1998). Temperatures between 73.4 and 78.8°F (23 and 26°C) can cause complete mortality in developing green sturgeon embryos, with upper limits for survival at around 62.6–64.4°F (17–18°C) (Van Eenennaam et al. 2005). Dolly Varden show decreased appetite above 60.8°F (16°C), and lethal temperatures are observed above 68.0°F (20°C) (Takami et al. 1997). A laboratory study of the early life-history stages of Pacific lamprey and western brook lamprey showed that the temperature for zero development for Pacific lamprey was 40.7°F (4.85°C), while for western brook lamprey it was 40.9°F (4.97°C); for both species, survival was greatest at 64.4°F (18°C) and lowest at 71.6°F (22°C), with abnormalities in the larval stage greatest in the 71.6°F (22°C) treatment for both species (Meeuwig et al. 2005).

Marine species also show preferences for certain temperature ranges. In a study of Puget Sound lingcod, Cook et al. (2005) showed that the optimal temperature for egg incubation was 48°F (9°C), and deformities were observed at 54°F (12°C) and 43°F (6°C). They also observed no hatch success at 59°F (15°C). Ocean surveys off the California coast showed that Pacific hake eggs were found most often in the 164- to 492-ft (50- to 150-m) depth range, and average temperatures for developing eggs were 50.4–52.3°F (10.2–11.3°C) (Moser et al. 1997). Finally, the upper limit of temperatures for Pacific cod in Nanao Bay (Japan) was 53.6°F (12°C) (Morioka and Kuwada 2002).

Elevated water temperatures can impair adult migration. Adult migration blockages occur consistently when temperatures exceed 69.8–71.6°F (21–22°C) (Poole and Berman 2001a). Thermal barriers to migration can isolate extensive areas of potentially suitable spawning habitat and contribute to prespawning mortality. Elevated temperature regimes also affect salmonid species by altering behavior and reducing resistance to disease and toxic substances. Studies have indicated that under chronic thermal exposure conditions, susceptibility of aquatic organisms to toxic substances may increase. Because elevated temperatures increase metabolic processes, gill ventilation also rises proportionately (Heath and Hughes 1973). Black et al. (1991) showed that an increase in water flow over the gills that results from increased gill ventilation at increased temperature resulted in the rapid uptake of toxicants, including metals and organic chemicals, via the gills. Salmonids also become more susceptible to infectious diseases at elevated temperatures (57–68°F [14–20°C]) because immune systems are compromised (Harrahy et al. 2001), while bacterial and viral activity is accelerated (Tops et al. 2006). In nearshore areas where temperature (as well as pollutant levels) may be elevated, the combined effect of thermal conditions and water pollution may be a primary driver of salmonid decline.

Considerably less research exists defining thermal criteria for invertebrates, although marine invertebrates can generally withstand higher temperatures. Gagnaire et al. (2006) noted that elevated temperatures caused blood cell mortality in Pacific oysters but not until temperatures exceeded 104°F (40°C), which is unlikely even in altered settings. In studies on northern abalone, optimal growth rates were found between 44.6 and 62.6°F (7 and 17°C) (Hoshikawa et al. 1998), with significant mortality at 32.9°F (0.5°C) and 79.7°F (26.5°C) (Paul and Paul 1998). It is unclear, however, what sublethal effect(s) may be significant with invertebrate populations. In general, an altered temperature regime will result in blocked migrations, increasing the chances of infection, deformities in developing eggs, stress, and mortality of several HCP species.

7.1.6.2 Altered Dissolved Oxygen

Dissolved oxygen (DO) content is critical to the growth and survival of all 52 HCP species. The amount of oxygen dissolved in water depends on temperature, physical mixing, respiration, photosynthesis, and, to a lesser degree, atmospheric pressure. These parameters can vary diurnally and seasonally and depend on activities such as oxygen inputs from daytime photosynthesis and nighttime plant respiration processes that deplete dissolved oxygen levels. Dissolved oxygen concentration is temperature dependent; as temperatures rise, the gas-absorbing capacity of the water decreases and the dissolved oxygen saturation level decreases. Reduced dissolved oxygen levels can be due to increased temperature (Snoeyink and Jenkins 1980), organic or nutrient loading (Ahearn et al. 2006), increased benthic sedimentation (Welch et al. 1998), or chemical weathering of iron and other minerals (Schlesinger 1997).

Depressed dissolved oxygen is associated with increased in biochemical oxygen demand (BOD) through eutrophication brought about by increased nutrient loading (Pickett 1997). Eutrophication is characterized by elevated primary production and associated with elevated respiration and decomposition of algae (Rabalais et al. 1996; Rabalais et al. 2001). In eutrophic systems, nighttime respiration drives down dissolved oxygen to levels that would adversely affect many of the HCP species. For example, depressed benthic oxygen levels in Hood Canal (Washington) have been associated with spot shrimp decline (Peterson and Amiotte 2006). The details of impacts from increased nutrients are discussed in Section 7.1.6.6 (*Altered Nutrient Loading)* below.

In the presence of dams, water releases may be not only cooler, but lower in dissolved oxygen from stratification in the reservoir (no mixing), no photosynthesis, and consumption of organic matter (Bednarek 2001). As a result, problems for some organisms, such as bull trout (Marshall et al. 2006), that occupy tailwaters can result. In other situations, tailwaters may be supersaturated in dissolved oxygen due to rapid drawdown (and therefore high velocities) and aeration through hydropower discharges (Bednarek 2001). Supersaturation has also been observed during dam removals (Wik 1995). While oxygen is the primary gas in supersaturated water, gas bubble disease involves other gasses as well, including nitrogen.

Supersaturation of water with dissolved oxygen has been documented to cause problems for several HCP species. In laboratory studies, exposure of white sturgeon larvae to 131 percent saturated dissolved oxygen water for 13 days caused 50 percent mortality (Counihan et al. 1998). Gas bubble disease has been documented from high dissolved oxygen as well, but this effect is temporary and overall populations are not always affected (Wik 1995); however, onset can be rapid (within 15 minutes at 131 percent dissolved oxygen water) (Counihan et al. 1998). Furthermore, increases in nitrogen from supersaturation may also lead to gas bubble disease.

7.1.6.2.1 Direct and Indirect Effects

Juvenile salmon are highly sensitive to low dissolved oxygen concentrations (USFWS 1986) and, consequently, are among the more vulnerable HCP species with regard to dissolved oxygen impairment. Salmon generally require dissolved oxygen levels of greater than 6 ppm for optimal survival and growth, with lethal 1-day minimum concentrations of around 3.9 ppm (Ecology 2002). Different organisms at different life-history stages require different levels of dissolved oxygen to thrive. Tolerance for low oxygen levels varies across species as well. For example, pygmy whitefish can withstand dissolved oxygen conditions below 5 ppm (Zemlak and McPhail 2006). Juvenile white sturgeon showed slower growth rates under hypoxic conditions (58 percent saturation) due to decreased food and oxygen consumption (Cech and Crocker 2002), and hypoxia also decreased their swimming activity (Crocker and Cech 1997). Table 7-5 lists the minimum recommended dissolved oxygen concentrations for salmonids and stream-dwelling macroinvertebrates (Ecology 2002). The dissolved oxygen thresholds presented in this table were derived from more than 100 studies representing over 40 years of research.

It should be noted that recommendations are presented in Table 7-5 for dissolved oxygen thresholds in categories other than lethality. Fish are motile organisms and, where possible, avoid dissolved oxygen levels that would cause direct mortality. However, this avoidance behavior in and of itself can affect fishes. Stanley and Wilson (2004) found that fish aggregate above the seasonal hypoxic benthic foraging habitat in the Gulf of Mexico, while Eby et al. (2005) found that fish in the Neuse River estuary (North Carolina) were restricted by hypoxic zones to shallow, oxygenated areas, where in the early part of the summer about one-third fewer prey resources were available. Studies such as these reveal how dissolved oxygen can change fish distributions relative to habitat, and potentially exclude fishes from reaching foraging and rearing areas. Sublethal dissolved oxygen levels can also cause increased susceptibility to infection (Welker et al. 2007) and reduced swim speeds (Ecology 2002), both of which may cause indirect impacts on HCP fish species.

Little consensus exists concerning low dissolved oxygen criteria for macroinvertebrates, and tolerances to hypoxic conditions that are taxonomically specific. Many invertebrates are adapted to live in benthic low-energy environments where dissolved oxygen concentrations are naturally low; consequently, these organisms can withstand hypoxic conditions. Kaller and Kelso (2007) found benthic macroinvertebrate density, including mollusks, greatest in low dissolved oxygen areas of a Louisiana wetland, while a literature review by Gray et al. (2002) found that in marine environments, invertebrates were not affected by low dissolved oxygen until concentrations fell below 1–2 ppm. Benthic dissolved oxygen levels can seasonally drop below this threshold in productive systems that receive high BOD loadings. For instance, depressed benthic dissolved oxygen levels in Hood Canal, Washington, have been associated with spot shrimp decline

(Peterson and Amiotte 2006). This dissolved oxygen decline has in turn been linked to BOD loadings from leaking or improperly functioning on-site wastewater disposal systems. These conditions in Puget Sound highlight the importance of reducing anthropogenically generated BOD.

Table 7-5. Summary of recommended dissolved oxygen levels for full protection[a] of salmonid species and associated macroinvertebrates.

Life-history Stage or Activity	Oxygen Concentration (ppm)	Intended Application Conditions
Incubation through emergence	≥9.0–11.5 *(30 to 90-DADMin)* and No measurable change when waters are above 52°F (11°C) (weekly average) during incubation.	Applies throughout the period from spawning through emergence. Assumes 1-3 ppm will be lost between the water column and the incubating eggs.
Growth of juvenile fish	≥8.0–8.5 *(30-DADMin)* and ≥5.0-6.0 *(1-DMin)*	In areas and at times where incubation is not occurring.
Swimming performance	≥8.0-9.0 *(1-DMin)*	Year-round in all salmonid waters.
Avoidance	≥5.0-6.0 *(1-DMin)*	Year-round in all salmonid waters.
Acute lethality	≥3.9 *(1-DMin)* ≥4.6 *(7 to 30-DADMin)*	Year-round in all salmonid waters.
Macroinvertebrates *(stream insects)*	≥8.5-9.0 *(1-DMin or 1-DAve)*	Mountainous headwater streams.
	≥7.5-8.0 *(1-DMin or 1-DAve)*	Mid-elevation spawning streams.
	≥5.5-6.0 *(1-DMin or 1-DAve)*	Low-elevation streams, lakes, and nonsalmonid waters.
Synergistic effect protection	≥8.5 *(1-DAve)*	Year-round in all salmonid waters to minimize synergistic effect with toxic substances.

Source: Ecology 2002.

[a] Full protection = approximately less than 1 percent lethality, 5 percent reduction in growth, and 7 percent reduction in swim speed.

1-DMin = annual lowest single daily minimum oxygen concentration.

1-DAve = annual lowest single daily average concentration.

7-, 30-, or 90-DADMin = lowest 7-, 30-, or 90-day average of daily minimum concentrations during incubation period, respectively.

7.1.6.3 Altered Suspended Solids and Turbidity

Several of the studies cited in this section present information in turbidity level units in the place of suspended sediment concentrations to infer effects thresholds. Turbidity is commonly used as a surrogate for suspended sediment concentrations, but the relationship between these measures is site specific. Where available, the equivalent suspended sediment concentration is provided, otherwise the turbidity value is provided. Because this complicates the interpretation of the information, a brief discussion of the relationship between turbidity and suspended sediment concentrations is provided here.

The International Standards Organization (ISO) defines turbidity as the "reduction of transparency of a liquid caused by the presence of undissolved matter" (Lawler, 2005), as measured by turbidimetry or nephelometry. Turbidity can be caused by a range of suspended particles of varying origin and composition. These include inorganic materials like silt and clay, and organic materials such as tannins, algae, plankton, micro-organisms and other organic matter. The term "suspended sediments" refers to inorganic particulate materials in the water column. Suspended sediments can range in size from fine clay to boulders, but the term applies most commonly to suspended fines (i.e., sand size or finer material). Because suspended sediments are a component of turbidity, turbidity is commonly used as a surrogate measure for this parameter. However, the accuracy of the results is dependent on establishing a clear correlation between turbidity and suspended sediment concentrations to account for the influence of organic materials. This correlation is site specific, given the highly variable nature of organic and inorganic material likely to occur in a given setting.

The presence of suspended solids and turbidity will result from any geomorphic change as well as various construction and maintenance activities associated with dams. Specific mechanisms of impact resulting in increased sediments include: (1) the release of clean water (see Section 7.1.2.1.3 [*Altered Sediment Transport*]), (2) alteration of flow regime (see Section 7.1.2.1.1 [*Altered Flow Regime*]), (3) alteration of natural sediment transport (see Section 7.1.2.1.3 [*Altered Sediment Transport*]), and (4) modification of riparian and aquatic vegetation (see Section 7.1.4 [*Riparian Vegetation Modifications*] and Section 7.1.5 [*Aquatic Vegetation Modifications*]), all of which have been discussed previously. In addition, dam removals can result in elevated levels of suspended sediments and turbidity.

Increased turbidity has a multitude of effects on HCP species. In general, the response of aquatic biota to elevated suspended solids concentrations is highly variable and dependent upon life-history stage, species, background suspended solids concentrations, and ambient water quality. Appendix A provides summary tables of the research that has been conducted on the effects of suspended solids on fish and invertebrates. The following sections provide detailed information on some of those findings.

7.1.6.3.1 Direct and Indirect Effects

Suspended sediments may affect fish by altering their physiology, behavior, and habitat, all of which may lead to physiological stress and reduced survival rates. For example, high levels of suspended solids may be fatal (lethal effects) to salmonids due to, for example, gill trauma, osmoregulation impairment, and changes in blood chemistry (Bash et al. 2001). Lower levels of suspended solids and turbidity may cause chronic sublethal effects, such as a loss or reduction in foraging capability, reduced growth, reduced resistance to disease, increased stress, and interference with cues necessary for orientation in homing and migration (Bash et al. 2001; Lloyd 1987; Newcombe and MacDonald 1991).

Lethal Effects

Although juveniles of many fish species thrive in rivers and estuaries with naturally high concentrations of suspended solids, studies have shown that the suspended solids concentration (as well as the duration of exposure) can be an important factor in assessing risks posed to salmonid populations (McLeay et al. 1987; Newcombe and MacDonald 1991; Servizi and Martens 1987). Lake and Hinch (1999) found suspended solids concentrations in excess of 40,000 ppm to elicit stress responses in juvenile coho salmon. Suspended solids concentrations this high would likely only be associated with construction activities. However, other studies have shown lethal effects at much lower concentrations.

Servizi and Martens (1991) exposed juvenile coho salmon to natural Fraser River suspended solids and found a 96-hour LC_{50} (the concentration at which a 50 percent population mortality was observed) of only 22,700 ppm. Using the identical apparatus and sediment source, juvenile sockeye salmon had a 96-hour LC_{50} of 17,600 ppm (Servizi and Martens 1987), and juvenile Chinook salmon had an LC_{50} of 31,000 ppm (Servizi and Gordon 1990). With lethal effects at concentrations as low as 17,600 ppm it is obvious that, for at least some species, the sublethal effects of suspended solids occur at even lower concentrations.

For white sturgeon, laboratory studies have shown that the survival of developing embryos was reduced to 5 percent in the presence of 0.2- to 0.8-inch (5- to 20-mm) thick layers of sediment compared to more than 80 percent survival in controls (Kock et al. 2006).

Thresholds for lethal effects on clams and eastern oysters have been reported, with negative impacts on eastern oyster egg development occurring at 188 ppm of silt (Cake 1983) compared to a 1,000 ppm threshold for hardshell clam eggs (Mulholland 1984). Suspended solids concentrations of <750 ppm allowed for continued larval development, but higher concentrations for durations of 10–12 days showed lethal effects for both clams and oysters. In addition, Ellis (1936) demonstrated that a silt accumulation of 0.24 to 0.98 inches (0.6–2.5 cm) resulted in mortality of 90 percent of test mussels in an artificial stream.

Collectively, these studies show no clear pattern of sublethal effects from elevated concentrations of suspended solids and thereby turbidity that could be generally applied across aquatic mollusks. This uncertainty is further complicated by the fact that many of the HCP invertebrate species are poorly studied. This indicates the need for directed studies on the sensitivity of these species before effects thresholds can be set. In the absence of this information, however, it is useful to consider that HCP invertebrates are all bottom-dwelling mollusks that have evolved to live in dynamic environments under conditions of variable turbidity. Therefore, sensitivity to turbidity-related stressors would be expected to occur only when conditions exceed the range of natural variability occurring in their native habitats.

Sublethal Effects

Studies on a variety of fishes, including sockeye and Chinook (Newcomb and Flagg 1983), coho, four-spine stickleback, cunner, and sheepshead minnow (Noggle 1978), attribute the observed chronic and acute impacts from high suspended solids to a reduced oxygen uptake (Wilber and

Clarke 2001). Fish must keep their gills clear for oxygen exchange. In the presence of high loadings of suspended solids, they engage a cough reflex to perform that function. Due to increased metabolic oxygen demand with increased temperatures and the need to keep respiration pathways free of sediments for oxygen uptake, increased temperature and reduced oxygen levels combine to reduce the ability of fish to cough and maintain ventilation rates. The stress induced by these conditions can lead to compromised immune defenses and reduced growth rates (Au et al. 2004). Sigler et al. (1984) observed reduced growth rates in juvenile steelhead and coho salmon at suspended solids concentrations as low as 100 ppm, while Servizi and Martens (1992) reported increased cough frequency in juvenile coho at concentrations of approximately 240 ppm.

Indirect effects on fish have been documented based on alteration of their food source. Suttle et al. (2004) observed that steelhead trout were affected by increased sediment because it caused a shift to burrowing macroinvertebrate taxa that then became unavailable as a food source.

Behavioral Effects

Aksnes and Utne (1997), Mazur and Beauchamp (2003), and Vogel and Beauchamp (1999) all report that suspended solids at sublethal concentrations have been shown to affect fish functions such as avoidance responses, territoriality, feeding, and homing behavior. Similarly, Wildish and Power (1985) reported avoidance of suspended solids by rainbow smelt and Atlantic herring at levels of 20 ppm and 10 ppm, respectively. However, it also appears that under certain circumstances, elevated suspended solids may actually benefit salmonids by providing cover (Gregory and Levings 1998) or triggering a sense of predation cover for salmonids (Gregory 1993). The study of Gregory (1993) indicated that when suspended solids concentrations exceeded 200 ppm, juvenile salmon increased their feeding rates while demonstrating pronounced behavioral changes in prey reaction and predator avoidance.

In studies of coho behavior in the presence of short-term pulses of suspended solids, Berg and Northcote (1985) found that territorial, gill flaring, and feeding behaviors were disrupted. At turbidity levels of between 30 and 60 nephelometric turbidity units (NTUs), social organization broke down, gill flaring occurred more frequently, and only after a return to a turbidity of 1–20 NTUs was the social organization reestablished. Similarly, feeding success was also found to be linked to turbidity, with higher turbidity levels reducing prey capture success.

Finally, in a study of dredging impacts on juvenile chum in Hood Canal, Salo et al. (1980) found that juvenile chum salmon showed avoidance reactions to even low suspended solids levels ranging from 2–10 ppm above ambient concentration. However, in related laboratory tests, Salo et al. (1980) found that avoidance was not observed until a concentration of 182 ppm was reached. In general, these behavioral thresholds vary across species and life-history stages.

Consistent with their early reliance on nearshore estuarine habitats that exhibit relatively high turbidities compared to pelagic or freshwater habitats, juvenile chum are classified as turbidity tolerant when compared to other fishes (Salo et al. 1980).

Habitat Effects

Increased turbidity is known to adversely affect submerged aquatic vegetation (Parkhill and Gulliver 2002; Terrados et al. 1998) such as eelgrass (Erftemeijer and Lewis 2006), which is associated with important rearing habitats for a suite of marine fishes including Pacific cod, Pacific salmon, rockfish, Pacific herring, and walleye pollock (Nightingale and Simenstad 2001a; Simenstad et al. 1999). In a study of the impact of sedimentation on seagrass in southeast Asia, Terrados (1998) noted that seagrass species richness and community leaf biomass declined sharply with a 15 percent increase in clay content of the sediments. Numerous studies have shown increased biomass of invertebrate (Cardoso et al. 2007; Seitz et al. 2005) and vertebrate species (Ferraro and Cole 2007; Pihl et al. 2006) within seagrass beds; thus, sedimentation-related negative impacts on seagrass arising from the construction or presence of flow control structures would likely affect the HCP species by decreasing the available nearshore habitat.

Burial of those invertebrate species having limited motility can lead to organism mortality as a direct effect of increased suspended sediments. Burial of invertebrate species occurs most frequently during the construction phase of a project. Limpets in intertidal habitat are affected by burial and interference with their feeding activity. In a field study in the UK, grazing by limpets (*Patella vulgata*) was decreased by 35 percent after the addition of fine sediment layers as little as 0.04 inch (1 mm) thick, with mortality and inhibition of feeding at higher levels of fine sediment (Airoldi and Hawkins 2007). The burial of mollusks and their subsequent stress or mortality resulting from partial and complete burial have been addressed empirically (Hinchey et al. 2006). Results of these studies indicate that species-specific responses vary as a function of motility, living position, and inferred physiological tolerance of anoxic conditions. Mechanical and physiological adaptations contribute to this tolerance. Olympia oysters have been shown to be intolerant of siltation and do best in the absence of fine-grained materials (WDNR 2006b).

As with dissolved oxygen, invertebrates tend to thrive across a wide range of suspended solids concentrations. Negative impacts on eastern oyster egg development have been shown to occur at 188 ppm total suspended solids (Cake 1983). Hardshell clam eggs appear to be more resilient, with egg development affected only after total suspended solids concentrations exceeded 1,000 ppm (Mulholland 1984). Mulholland (1984) showed that suspended solids concentrations of <750 ppm allowed for continued larval development, but higher concentrations for durations of 10–12 days showed lethal effects for both clams and oysters.

When suspended solids concentrations rise above the filtering capacities of bivalves, their food becomes diluted (Widdows et al. 1979). Studies have shown that the addition of silt, in relatively low concentrations in environments with high algal concentrations, can be marked by the increased growth of mussels (Kiorboe et al. 1981), surf clams (Mohlenberg and Kiorboe 1981), and eastern oysters (Urban and Langdon 1984). Bricelj and Malouf (1984), however, found that hardshell clams decreased their algal ingestion with increased sediment loads, and no growth rate differences were observed between clams exposed to algal diets alone and clams with added sediment loads (Bricelj et al. 1984). Urban and Kirchman (1992) reported similarly ambiguous results concerning suspended clay. Suspended clay (20 ppm) interfered with juvenile eastern oyster ingestion of algae, but it did not reduce the overall amount of algae ingested. Grant et al. (1990) found that the summer growth of European oysters was enhanced at low levels of sediment resuspension, and inhibited with increased deposition. It was hypothesized that the chlorophyll in suspended solids may act as a food supplement that could enhance growth, but higher concentrations of suspended solids may dilute planktonic food resources, thereby suppressing food ingestion. Changes in behavior in response to sediment loadings were also noted for soft-shelled clams in sediment loads of 100–200 ppm, with changes in their siphon and mantles over time (Grant and Thorpe 1991).

7.1.6.4 Increases in Contaminated Sediment

A cumulative impact on water quality occurs when increased suspended sediments are contaminated. Sediments trapped behind dams are usually fine, and small particles tend to adsorb contaminants (Murakami and Takeishi 1977). Contaminated sediments are an issue with dam projects when proposed construction occurs near contaminated sites. Dredging and construction equipment activity can contribute to benthic resuspension and increase the availability of contaminated sediments for biotic assimilation. Contaminated sediments are of particular concern due to the risk of contaminant transport and exposure posed to aquatic organisms through bioaccumulation and biomagnification in the freshwater and marine food web. When considering a dam removal, it is important to assess the condition of accumulated sediments before deconstruction in order to minimize the release of both contaminants and suspended sediments. For example, during an accidental dam breach in New York, contaminated sediments caused an increase in the levels of polychlorinated biphenyls (PCBs) downstream (Shuman 1995).

7.1.6.4.1 Direct and Indirect Effects

Many contaminants can accumulate on sediments including pesticides, PCBs, endocrine disruptors, polycyclic aromatic hydrocarbons (PAHs), metals, and nutrients (Bednarek 2001). These contaminants may lead to reproductive problems and abnormalities in many of the HCP species. For example, in the Colorado River, Feist et al. (2005) showed that plasma androgens and gonad size in male white sturgeon were negatively correlated with total DDT, total pesticides, and PCBs. In a study of Columbia River white sturgeon, Burner and Rein (2002) measured the occurrence of physical deformities, which included an additional row of lateral scutes on both sides of the fish and misshapened fins. Although they could not show a clear causal relationship, the authors inferred that these deformities might be the result of organics in the sediments, which are known to be harmful to aquatic organisms (Burner and Rien 2002).

Studies have shown that fishes and invertebrates exposed to contaminants may bioaccumulate and concentrate trace pollutants to harmful levels. For example, large vessels (i.e., more than 82 ft [25 m] in length) are allowed to use tributyltin bottom paint, which is highly toxic to aquatic organisms. Studies have shown that tributyltin can biomagnify through algae, invertebrate, and vertebrate species (Mamelona and Pelletier 2003). This accumulation of contamination in biota occurs after contaminants are passed between two or more trophic levels. This transfer between multiple trophic levels (i.e., prey to predator) is known as trophic transfer.

Different contaminants have different biomagnification potential, and many pollutants can be found in contaminated sediment of historically industrial or highly urbanized areas. Studies in the Pacific Northwest by Stein et al. (1995) and Johnson et al. (2007) have indicated that PCB and PAH concentrations in juvenile Chinook salmon tissue are highest in industrial areas (e.g., the Duwamish estuary, Columbia River). Consequently, construction associated with flow control structures in these areas should be accompanied by extensive sediment sampling to assess the potential for contaminant mobilization.

The number of potential contaminants associated with sediments is vast and highly dependant on site-specific conditions. The examples above illustrate how organic contaminants can bioaccumulate in fish and invertebrates. Other contaminants, such as metals, PAHs, and nutrients, are discussed elsewhere in this white paper. Metal toxicity is discussed in detail in Section 7.4 (*Outfalls*) because metals are often found in stormwater runoff and are common to this subactivity type. Impacts from PAHs are discussed in Section 7.1.6.7 (*Introduction of Toxic Substances*). Finally, impacts from nutrient enrichment are discussed in detail in Section 7.1.6.6 (*Altered Nutrient Loading*).

7.1.6.5 Altered pH Levels

The pH of fresh and salt water normally ranges from 6.5–8.5 (Schlesinger 1997). The construction of dams or other flow control structures using concrete can affect the pH of surrounding waters if the uncured concrete is allowed to contact the receiving water body. Uncured concrete can dissolve in water and, depending on the temperature, can raise the pH level to as high as 12, which is far outside the livable range for all of the HCP species (Ecology 1999). This impact will be greatest during construction when concrete wash off and slurries come into contact with water (Dooley et al. 1999), but once construction is complete concrete may still affect the surrounding environment. Curing concrete surfaces can exhibit pH values as high as 13 during the 3 to 6 months it takes for concrete to cure underwater (Dooley et al. 1999). This elevated pH prevents attached macroalgae growth during this period.

Altered pH from curing concrete will increase pH to levels that can affect fish, invertebrates, and their food. But this effect is localized and—as stated above—should last no more than 6 months. Consequently, it is estimated that this impact mechanism will be most significant for large projects in areas with poor water circulation.

7.1.6.5.1 Direct and Indirect Effects

Fish have adapted to the ambient pH levels of their particular habitat and tend to have narrow ranges of pH tolerance. The effects of high pH levels outside of their tolerance range can include death; damage to gills, eyes, and skin; and an inability to excrete metabolic wastes (DFO 2007). When ambient conditions are characterized by elevated ammonia and pH, ammonia toxicity in fish can occur because the organisms have difficulty excreting ammonia waste through their gills. At ambient ammonia concentrations of 5 ppm, the mortality of tambaqui (*Colosoma macropomum*; also known as pacu), a neotropical fish, increased from 0 to 15 to 100 percent at a pH of 7, 8, and 9, respectively (de Croux et al. 2004). Consequently, if ammonia concentrations are elevated due to waste dumping from recreational vessels or from upland sources, the toxicity may be compounded by elevated pH from construction activities.

Changes in pH level, particularly rapid changes that outpace physiological adaptation, can adversely affect fish. When levels exceed or fall below tolerance thresholds, effects can progress rapidly—from stress and behavioral avoidance to injury and death. For example, Wagner et al. (1997) studied the effects of elevated pH on rainbow trout and found that levels above 8.4 caused an increase in glucose and cortisol levels, physiological responses indicative of stress. pH levels above 9.3 caused mortality. These thresholds should not be viewed as absolutes, however, as the sensitivity of trout to elevated pH, and presumably other freshwater species who live in environments with variable water chemistry, is dependent on acclimatization conditions.

Even in the absence of observable effects, nonlethal physiological responses to altered pH levels can have important implications. For example, white sturgeon sperm cells demonstrated decreased sperm motility after exposure to pH levels below 7.5, even when conditions returned to optimal pH levels (Ingermann et al. 2002). This suggests that even temporary exposure to changes in pH during critical life-history periods could affect survival and fitness in HCP fish species.

Alterations in pH can also affect invertebrates. The majority of research on the effect of pH on invertebrates is related to the impact of acidification on abundance and diversity; consequently, there is little research on the impact of elevated pH on invertebrates. In a study of the freshwater Malaysian prawn, Cheng and Chen (2000) noted a 38 percent decrease in haemocyte (invertebrate blood cell) count when pH dropped below 5 or rose above 9. In another study, Bowman and Bailey (1998) found that zebra mussels have an upper pH tolerance limit of 9.3 through 9.6. From these studies, it can be assumed that pH levels that exceed a pH of between 9 and 10 will have a negative impact on invertebrate HCP species. As indicated above, pH levels on and around curing concrete can exceed this pH threshold and thus there is the potential for impact on local invertebrate communities.

7.1.6.6 Altered Nutrient Loading

When dams are present, the most common cause of increased nutrient loading occurs when the bottom layer of upstream reservoirs is released because this water is often high in nutrients (Camargo et al. 2005; Palmer and Okeeffe 1990; Teodoru and Wehrli 2005). Furthermore, any activity that decreases in-channel processing of nutrients or that reduces riparian buffer widths will contribute to the increased export of nutrients to downstream receiving waters, potentially affecting many of the HCP species. Numerous studies have shown that wide stream buffers are effective at attenuating nutrients (Feller 2005; Mayer et al. 2005), herbicides (Gay et al. 2006), and sediment loading (Jackson et al. 2001). Riparian vegetation retards overland flow, promotes infiltration, and assimilates shallow groundwater nutrients. When this vegetation is removed through any HPA-permitted activity, nutrients and pollutants will be more efficiently transported from upland sources to downgradient water bodies. Forested buffers can effectively remove nutrients in shallow groundwater. In a study of a forested buffer in Alabama, a 33-ft (10-m) buffer reduced the groundwater nitrate concentration by 61 percent (Schoonover and Williard 2003). In a subsequent study of a forested wetland buffer, a buffer averaging 125 ft (38 m) in width reduced the nitrate concentration by 78 percent and total phosphorus by 66 percent (Vellidis et al. 2003).

Changes in groundwater–surface water interactions and hyporheic exchange from the loss of riparian vegetation will reduce nutrient cycling in the river. Aquatic vegetation takes up nutrients in river water, so the loss of algae from construction and an increase in sedimentation will increase nutrient loading downstream. The pathways leading to altered nutrient loading for these mechanisms of impact are described in detail in Section 7.1.4 (*Riparian Vegetation Modifications*) and Section 7.1.5 (*Aquatic Vegetation Modifications*). Loss of channel complexity resulting from a loss in LWD recruitment also influences nutrient loading.

7.1.6.6.1 Direct and Indirect Effects

Increased nutrient loading may be beneficial to fish in pristine upland systems. When riparian canopies are opened, increased photosynthetic active radiation reaches the channel, temperatures increase, and nutrient loading increases. These alterations can increase macroinvertebrate abundance and biomass as well as algal biomass (Fuchs et al. 2003; Hetrick et al. 1998). However, the cumulative effect of increased nutrient loading contributes to eutrophication in downstream receiving waters. Eutrophication refers to the increase in nutrient pollution to receiving waters and has been identified as a major source of environmental degradation of receiving waters throughout Washington State (Nelson et al. 2003; Pickett 1997). Eutrophication occurs when limits to vegetative growth are reduced. In Washington, the primary limiting nutrient in fresh water is phosphorus. This is due to the fact that abundant iron in freshwater systems binds with phosphorus (P) and reduces the availability of P for biotic assimilation. In marine waters, however, nitrogen is often the limiting nutrient. In marine waters, iron preferentially binds with sulfide, and the associated P is released; this creates conditions of nitrogen limitation (Blomqvist et al. 2004).

When nutrient limitations are eliminated, vegetative growth increases. This process accelerates carbon fixation; the additional carbon loading to the aquatic system increases respiration as heterotrophs use carbon for energy. Through the process of carbon oxidation, oxygen is converted to carbon dioxide (CO2) and ambient dissolved oxygen levels decrease. Eutrophication-induced hypoxia is a nationwide problem (Scavia and Bricker 2006). The ramifications of low dissolved oxygen on HCP species are addressed above and in Section 9 (Potential Risk of Take).

In Washington State, low dissolved oxygen episodes in Hood Canal have resulted in widespread fish and invertebrate kills (Peterson and Amiotte 2006). These low dissolved oxygen episodes have been linked to excess carbon loading due to nutrient enrichment. The resultant algal blooms may affect not only dissolved oxygen levels but also, if certain species flourish, contribute to paralytic shellfish poisoning (Horner 1998). Nutrient loading may increase in the presence of dams and other flow control structures through the loss of riparian and aquatic vegetation, the introduction of nutrients from construction and maintenance activities, and changes in groundwater–surface water interactions. These pathways can all result in eutrophication of the receiving waters.

The end result of the eutrophication process is a reduction in the dissolved oxygen concentration; therefore, the indirect impacts from altered nutrient loading is discussed in detail in Section 7.1.6.2 (*Altered Dissolved Oxygen*).

7.1.6.7 Introduction of Toxic Substances

Sediments accumulate behind dams as a result of lowered stream velocities, thereby allowing sediments to settle and deposit in the reservoir. In areas where contamination from organics, pesticides, and metals occurs, these will adsorb to sediments and accumulate behind dams. The release of these contaminated sediments may occur during maintenance activities or during dam removal. Dams can result in the introduction of toxic substances through the resuspension of contaminated sediments and accidental fuel and chemical spills. This will occur primarily during construction and maintenance activities. Another source of toxic chemicals is accidental spills from increased recreational vessel use encouraged by the creation of the impoundment upstream of a dam. The introduction of toxic substances from recreational uses induced by dam development is potentially important because these small chronic sources occur at a greater frequency than for infrequent construction and maintenance activities.

This section focuses on petroleum-based contaminants such as fuel, oil, and hydraulic fluids, which contain PAHs. This represents another group of potentially harmful chemicals that can result in direct and indirect impacts on HCP species. For a discussion of the direct and indirect effects from metals, the reader is directed to Section 7.4.6.1 (*Metals Toxicity*) under Section 7.4 (*Outfalls*).

7.1.6.7.1 Direct and Indirect Effects

PAHs can be acutely toxic to salmonids at high levels of exposure and may also cause chronic lethal and sublethal effects on aquatic organisms (Hatch and Burton 1999). Misitano et al. (1994) exposed larval surf smelt to Puget Sound (Eagle Harbor) sediments containing high concentrations of PAHs (284 to 464 ppm sediment) and observed 100 percent mortality after 96 hours of exposure. After diluting the sediments and repeating the experiments, the researchers found that the larvae which did not expire within 96 hours suffered from decreased growth rates. In a study of Pacific herring, mature herring and developing embryos were both exposed to weathered crude oil. The herring showed increased levels of PAHs, and progeny of mature herring were not affected after a 16-day exposure, whereas the developing embryos were (Carls et al. 2000). The authors discussed that gametes were given “parental protection” because exposure and accumulation was direct to the adults, and indirectly passed to the ova. On the other hand, developing embryos exposed to PAHs showed higher concentrations, higher incidence of yolk-sac edema, and decreases in larval length. As a result of this study, the authors concluded that Pacific herring embryos were more sensitive to exposure to PAHs than gametes.

7.1.6.8 Altered Salinity

Altered flow regimes in the presence of dams will alter the volume and timing of freshwater inputs to estuarine ecosystems. Altered salinity gradients can result in migration difficulties, alteration in the mixing of freshwater and brackish water, and community composition changes. These impacts and their effect on HCP fish and invertebrate species are described in detail in Section 7.6.6.6 (*Altered Salinity*) in Section 7.6 (*Tide Gates*).

7.2 Weirs

The term "weir" is broadly applied to a variety of subactivity types. For the purpose of this white paper, weirs are structures that partially or fully span a channel and are used for flow control and/or water diversion. Weirs used for flow control purposes can include both full channel spanning structures that act like a small dam, and structures that partially span the channel. Full channel spanning weirs will create an impoundment upstream of the structure and alter flow conditions downstream. Some types of intake and diversion structures (discussed in Section 7.5 [*Intakes and Diversions*]) may incorporate partial channel spanning weir structures that do not form upstream impoundments.

This white paper does not explicitly address the effects of weir structures used to control fish passage or to provide habitat or channel modifications, although the effects of these types of structures on the environment may be similar in some cases to the weirs described herein. Weirs constructed for fish passage management include both permanent and temporary structures. Temporary weirs are often installed to facilitate the counting of adult fish returning to spawning grounds and have relatively modest effects on the aquatic ecosystem. Permanent weirs are often used for similar purposes, as well as to prevent fish passage to upstream areas consistent with specific management objectives. For example, hatchery weirs are commonly used to control the upstream passage of hatchery fish (while serving the dual purpose of diverting operational water into the facility). Weir type structures are also used in specific circumstances for the sole purpose of preventing the invasion of non-native species into habitats where sensitive native species are present. Depending on their size and configuration, this type of structure may have effects rivaling those imposed by flow control oriented weirs. The effects of fish passage weirs are addressed in the Fish Passage white paper (Herrera 2007a).

Habitat structures composed of logs and/or rock used for grade control and habitat enhancement are also commonly referred to as weirs (e.g., log K-weirs). The effects of large woody debris placement and in-channel habitat creation are discussed in the Habitat Modifications white paper (Herrera 2007e).

Mechanisms of impact associated with weirs include the same six general impact pathways as described previously for dams: construction and maintenance activities, hydraulic and geomorphic modifications, ecosystem fragmentation, riparian vegetation modifications, aquatic vegetation modifications, and water quality modifications. In terms of these mechanisms of impact, weirs act similar to a small-scale dam, so their impact mechanisms and submechanisms are similar, but generally to a lesser degree. As a result, extensive cross-referencing is used in this section, and the complete suite of submechanisms of impact are not presented as independent subsections. Where differences between dams and weirs exist, additional information is presented.

7.2.1 Construction and Maintenance Activities

Construction and maintenance activities from weirs will have similar impacts on fish and invertebrates as dams. Activities associated with weir construction and repair pose the risk of increasing underwater noise levels, increasing suspended solids, removing or disturbing aquatic and riparian vegetation, disturbing banks and shorelines, and releasing toxic substances from construction materials and/or construction equipment to fresh and marine waters. Construction activities may also include filling and dredging that can entrain organisms or permanently remove habitat for burrowing and benthic animals. For details on the direct and indirect effects from construction and maintenance activities on fish and invertebrates and their habitats see Section 7.1.1 (*Construction and Maintenance Activities*) in Section 7.1 (*Dams*).

7.2.2 Hydraulic and Geomorphic Modifications

The presence of a weir influences the hydraulic and geomorphic properties of a riverine system, which can have direct and indirect effects on HCP species. In addition, weirs are typically associated with tide gates in marine environments. For the hydraulic and geomorphic impacts from these types of weirs, see Section 7.6 (*Tide Gates*). In general, hydraulic and geomorphic impacts include five submechanisms of impact that are identified for analysis in this white paper: (1) altered flow regime, (2) altered sediment transport, (3) altered channel geometry, (4) altered substrate composition, and (5) altered groundwater–surface water interactions. These submechanisms result in identical effects on fish and invertebrates as previously described for dams. For example, flow over weirs will increase turbulence below structures and increase local velocities, making fish passage difficult (Baker 2003). Sea lampreys have been observed migrating over weirs with short bursts of movement following by extended resting periods (Quintella et al. 2004). The sea lampreys seemed affected by increasing fatigue, which the authors attributed to initiating a new burst of movement without fully recovering from the previous efforts.

In addition, weirs represent drops in channel elevation, which can alter channel slopes. Abrupt changes in slope can alter sediment transport and represent migration barriers for fish. In a study of fall heights from weirs on movements with the common bully (*Gobiomorphus cotidianus*) and adult and juvenile inanga (*Galaxias maculatus*), researchers showed that both species were restricted by falls of 0.4 inches (10 cm), and the passage of adult inanga was restricted by falls of 0.8 inches (20 cm) (Baker 2003). Atlantic salmon in the Pau River (France) were able to pass over weirs of 59.1 inches (1.5 m) in height but had difficulty passing weirs of 98.4 inches (2.5 m) in height (Chanseau et al. 1999).

For weirs, the direct and indirect effects on fish and invertebrates from altered flow regime, altered channel geometry, altered sediment transport, altered substrate composition, and altered groundwater–surface water interactions are the same as for dams and are summarized in Section 7.1.2 (*Hydraulic and Geomorphic Modifications*) under Section 7.1 (*Dams*).

7.2.3 Ecosystem Fragmentation

The presence of a weir affects HCP species through direct fragmentation of ecosystems. Longitudinal connectivity is the most documented type of ecosystem fragmentation in the literature. Similar to alterations in slope and velocity discussed above, weirs result in delayed migrations of fish that need to navigate over them (Chanseau et al. 1999). In Denmark, Atlantic salmon and brown trout losses increased due to delayed migrations and increased predation while these fish were trying to negotiate weirs (Aarestrup and Koed 2003). A similar result was observed with brown trout in the Bidasoa River in Spain (Gosset et al. 2006). In Australia, radio-tagged fish were removed from a river after passing over (and under) weirs and were placed back downstream of the structures. When faced with passing the weir a second time, few of the fish did, with most trying to avoid it altogether (O'Connor et al. 2006). Finally, ecosystem fragmentation can lead to changes in genetic diversity. A study of the genetic variation within European grayling populations from the Skjern River (Denmark) showed that present-day grayling differed from historic stocks due to the drift of larvae downstream and restricted migration upstream (Meldgaard et al. 2003). However, some studies have shown that restricted upstream movements are dependant on fish size, with larger fish able to pass small weirs more easily than smaller fish (Baker 2003; Winter and Van Densen 2001). Finally, although not as effective as dams, weirs do have the potential to trap large woody debris in a river system.

For weirs, the direct and indirect effects on fish and invertebrates are the same as for dams and include altered longitudinal connectivity, altered river–floodplain connectivity, altered LWD transport and recruitment, and altered community composition. These effects are summarized in Section 7.1.3 (*Ecosystem Fragmentation*) under Section 7.1 (*Dams*).

7.2.4 Riparian Vegetation Modifications

Impacts from riparian vegetation modifications are the same for weirs as for dams. Namely, vegetation removal during construction and maintenance activities, or bank failures from altered sediment transport will lead to changes in riparian vegetation. Removal or disturbance of riparian vegetation during construction activities permitted under HPAs can expose HCP species to stressors caused by a variety of impact mechanisms, including:

- Altered shading, solar input, and ambient air temperatures
- Altered bank and shoreline stability
- Altered allochthonous inputs
- Altered groundwater–surface water interactions
- Altered habitat complexity
- Increased nutrient/pollutant loading.

These impact mechanisms and related ecological stressors, as well as direct and indirect impacts are described in Section 7.1.4 (*Riparian Vegetation Modifications*) under Section 7.1 (*Dams*).

7.2.5 Aquatic Vegetation Modifications

The submechanisms of aquatic vegetation modifications for weirs are the same as for dams: altered autochthonous production, altered habitat complexity, and altered nutrient/pollutant loading. Aquatic vegetation can be altered by changes in sediment transport (burial), scour from altered flow velocities, and loss by construction and maintenance activities. The effects on fish and invertebrates from aquatic vegetation modifications are described in Section 7.1.5 (*Aquatic Vegetation Modifications*), Section 7.1 (*Dams*).

7.2.6 Water Quality Modifications

Water quality modifications that occur as a result of weirs are the same as those impacts caused by dams. In general, weirs alter the following water quality variables: temperature, dissolved oxygen, suspended sediments (turbidity), pH levels, and nutrient and pollutant loading. Other water quality modifications such as increases in contaminated sediments, introduction of toxic substances, metal toxicity, and altered salinity are not common in the presence of weirs. Therefore, they are not be discussed here. However, if a weir structure is big enough to act like a dam and the impoundment is used for recreation, these mechanisms of impact might become more important. If this is the case, refer to Section 7.1.6 (*Water Quality Modifications*) under Section 7.1 (*Dams*).

Similar to a dam, water is slowed behind a weir, and pooled water increases in temperature and flows downstream. Temperature changes from a weir are likely smaller than for a dam because weirs generally have smaller impoundments and are often run-of-the-river structures. Similar to dams, flow over weirs can result in supersaturated dissolved oxygen concentrations due to high velocities (Baylar and Bagatur 2000). During construction and maintenance activities, suspended sediments will increase, and pH can be altered from the use of concrete. Finally, alteration of riparian and aquatic vegetation will alter groundwater–surface water interactions and alter nutrient cycling within the system.

A detailed discussion of the impacts on HCP species from water quality alterations is presented in Section 7.1.6 (*Water Quality Modifications*) in Section 7.1 (*Dams*).

7.3 Dikes and Levees

Dikes and levees are built to maintain flows within a confined channel for flood control purposes. In addition, they are sometimes used to convert estuarine habitat into agricultural fields or freshwater habitat (e.g., used on WDFW lands and federal wildlife refuges to provide waterfowl habitat/hunting areas). In some cases, dikes are built in tidal areas in conjunction with tide gates. For impacts resulting from these types of dikes see Section 7.6 (*Tide Gates*). This section addresses the direct and indirect impacts of dikes and levees on fish and invertebrates, their habitats, and ecological processes. In this white paper, the terms dikes and levees are used interchangeably.

The mechanisms of impact from dikes and levees are the same six general impact pathways discussed previously for dams and weirs: construction and maintenance activities, hydraulic and geomorphic modifications, ecosystem fragmentation, riparian vegetation modifications, aquatic vegetation modifications, and water quality modifications. Because many of the submechanisms of impact for dikes and levees are the same as for dams, this section contains extensive cross-referencing, and the complete suite of submechanisms of impact are not presented as independent subsections. Where differences between dams and levees exist, additional information is presented.

7.3.1 Construction and Maintenance Activities

Construction activities associated with dikes and levees will have impacts on fish and invertebrates similar to those described for dams. However, levees generally require less-intrusive construction activities. For example, pile driving would not be likely to occur because most construction is completed using earthen materials, including rock. Subsequently, underwater noise, while possible from equipment operation and materials placement, is likely less than for dam construction. Work area dewatering would also likely be minimized because most levees and dikes are constructed adjacent to rivers and wetland areas. However, increases in suspended sediment may be large due to the nature of using earthen materials for construction. Construction activities may also include filling and dredging, which can entrain organisms or permanently remove habitat for burrowing and for benthic animals. Finally, construction activities may remove or disturb aquatic and riparian vegetation, disturb banks and shorelines, and release toxic substances from construction materials and/or construction equipment into fresh and marine waters. For details on these impacts from construction and maintenance activities on fish and invertebrates and their habitats, see Section 7.1.1 (*Construction and Maintenance Activities*) in Section 7.1 (*Dams*).

7.3.2 Hydraulic and Geomorphic Modifications

Dikes and levees alter the hydraulic and geomorphic properties of the environment where they are located. In a riverine system, dikes and levees reduce a river's connection with its floodplain and increase peak flows (Liu et al. 2004). This can lead to habitat isolation and strand fish in isolated pools without connection to the mainstem, and prevent access to low velocity refuge areas (Bolton and Shellberg 2001). A complete discussion of altered floodplain connectivity is provided in Section 7.1.3.1.2 (*Altered River-Floodplain Connectivity*) under Section 7.1 (*Dams*).

Levees typically confine river flows to straightened channels, reducing channel sinuosity and altering channel geometry and sediment transport. In addition, this confined flow tends to have higher velocities and deeper water compared to conditions prior to construction, leading to increased erosion downstream (Bolton and Shellberg 2001). Impacts on fish and invertebrates from these mechanisms are described in Section 7.1.2 (*Hydraulic and Geomorphic Modifications*) in Section 7.1 (*Dams*).

In tidal marshes, impacts will be similar and include changes in channel geometry, sediment transport, and flow regime. These impacts are described in Section 7.1.2 (*Hydraulic and Geomorphic Modifications*) in Section 7.1 (*Dams*). In addition, due to their proximity to tidal areas, dikes located in nearshore sloughs and estuaries can lead to changes in wave energy, current velocities, and nearshore circulation. In a study of the Skagit River delta, dikes caused a reduction in tidal flushing, which increased sedimentation within the tidal area and reduced channel sinuosity (Hood 2004). Furthermore, loss of floodplain area to dikes prevents flood energy dissipation over the marsh surface, causing the mean channel width to increase and sinuosity to decrease (Hood 2004).

A detailed discussion of impacts on fish and invertebrates from altered wave energy and nearshore circulation is presented in Section 7.4.2.1 (*Submerged Outfalls*). In addition, sediment compaction occurs in disconnected tidal marshes from the increased oxidation of organic matter, which can potentially lead to water quality impacts (Anisfeld et al. 1999). These impacts are described in Section 7.3.6 (*Water Quality Modifications*).

7.3.3 Ecosystem Fragmentation

The presence of dikes and levees will affect HCP species through the direct fragmentation of ecosystems. Ecological connectivity is essential between riverine, floodplains, and riparian ecosystems (Stanford and Ward 1993). The main ecological connection altered in the presence of a dike or levee is the river–floodplain connection. After construction of a dike or levee, floodplain, lagoon, and tidal marsh habitats are lost (Hood 2004; Liu et al. 2004). These habitats are important for many of the HCP species. For example, after dikes were removed on the Salmon River (Oregon), juvenile fall Chinook salmon were observed using many regions of the restored tidal marsh (Bottom et al. 2005). A loss of floodplain habitat in China from the construction of dikes and levees resulted in a 74 percent reduction in habitat and caused declines in many types of plant species (Liu et al. 2004). A detailed discussion of the impacts on fish and invertebrates from a loss in lateral connectivity is presented in Section 7.1.3.1.2 (*Altered River–Floodplain Connectivity*) in Section 7.1 (*Dams*). For a discussion of the impacts from the loss of lagoon habitat, see Section 7.6.3 (*Ecosystem Fragmentation*) in Section 7.6 (*Tide Gates*).

In addition to floodplain habitat loss, the changes in hydrology and geomorphology could potentially lead to an increase in the invasion of exotic species. However, little information is available on this topic and represents a potential data gap.

7.3.4 Riparian Vegetation Modifications

Riparian vegetation may be modified through its removal and disturbance during construction and maintenance activities related to levees. In addition, changes in flow regime and altered sediment transport will impact riparian vegetation. Removal or disturbance of riparian vegetation during construction or activities from levee projects can expose HCP species to stressors caused by a variety of impact mechanisms. These mechanisms include:

- Altered shading, solar input, and ambient air temperature
- Altered bank and shoreline stability
- Altered allochthonous inputs
- Altered groundwater–surface water interactions
- Altered habitat complexity
- Increased nutrient/pollutant loading

These impact mechanisms and their related ecological stressors, as well as direct and indirect impacts are discussed in detail in Section 7.1.4 (*Riparian Vegetation Modifications*) in Section 7.1 (*Dams*).

In tidal areas, riparian zones serve similar functions as they do in riverine environments. Therefore, modification of marine riparian vegetation will likely result in impacts on several HCP species. A detailed discussion of the impacts from marine riparian vegetation modifications on HCP species is presented in Section 7.6.4 (*Riparian Vegetation Modifications*) in Section 7.6 (*Tide Gates*).

7.3.5 Aquatic Vegetation Modifications

Aquatic vegetation modifications can occur during levee construction as a result of equipment access, dredging, increased turbidity, and from alteration of hydrology. Any activity that mechanically removes or by other means affects aquatic vegetation will alter autochthonous production and habitat complexity, as well as reduce the sediment, nutrient, and pollutant retention and reduction capabilities of the aquatic system. Indirect impacts from the removal of aquatic vegetation include increased nutrient and pollutant loading to receiving waters, which could exacerbate eutrophic conditions (creating low oxygen conditions) and/or metals toxicity. A detailed discussion of the impacts on HCP species from nutrient loading is presented in Section 7.1.6.6 (*Altered Nutrient Loading)* in Section 7.1 *(Dams)*. In addition, the loss of aquatic vegetation could alter food webs through changes in energy cycling and available sources. For levees located in a riverine environment, details on impacts from aquatic vegetation modifications are provided in Section 7.1.5 (*Aquatic Vegetation Modifications*) in Section 7.1 *(Dams)*. In tidal areas, seagrasses have been linked to improved water quality. As an example, Moore (2004) noted decreased nutrient concentrations and turbidity levels in seagrass beds relative to areas outside the beds along the littoral zone of the Chesapeake Bay National Estuarine Research Reserve. For levees located in tidal marshes, details on the impacts on fish and invertebrates are presented in Section 7.6.5 (*Aquatic Vegetation Modifications*) in Section 7.6 (*Tide Gates*).

7.3.6 Water Quality Modifications

Water quality modifications resulting from dikes and levees are similar to those discussed previously for dams. In general, dikes and levees will alter temperature, dissolved oxygen, suspended sediments (turbidity), salinity, and nutrient and pollutant loading.

7.3.6.1 Altered Temperature Regime

Temperature will change as a result of altered hydraulics both in riverine and tidal systems. The impacts from altered temperatures on HCP species are described in Section 7.1.6.1 (*Altered Temperature Regime*) in Section 7.1 (*Dams*).

7.3.6.2 Altered Dissolved Oxygen

Dissolved oxygen depletion has been observed in diked tidal marshes. As a result of restricted tidal flows and inputs of nutrients and organic matter from adjacent wetlands, anoxia in the Herring River estuary (Massachusetts) occurs every summer (Portnoy 1991). This low oxygen environment is often accompanied by fish kills. For a detailed discussion of the impacts on HCP species from low dissolved oxygen, see Section 7.1.6.2 (*Altered Dissolved Oxygen*) in Section 7.1 (*Dams).*

7.3.6.3 Altered Suspended Solids and Turbidity

Sediment transport is altered by changes in the flow regime and as a result of sediment generated from construction and maintenance activities. The impacts on fish and invertebrates are described in Section 7.1.6.3 (*Altered Suspended Sediments and Turbidity)* in Section 7.1 (*Dams).*

7.3.6.4 Altered Nutrient Loading

Any activity that decreases the in-channel processing of nutrients or riparian buffer widths will increase the export of nutrients to downstream receiving waters, potentially affecting many of the HCP species. This includes those activities that modify riparian and aquatic vegetation. The pathways leading to altered nutrient loading from these mechanisms of impact are described in Section 7.1.4 (*Riparian Vegetation Modifications*) and Section 7.1.5 (*Aquatic Vegetation Modifications*), respectively, in Section 7.1 (*Dams).* Furthermore, a loss of channel complexity resulting from reductions in LWD also influences nutrient loading. A detailed discussion of the impacts on species from nutrient loading is presented in Section 7.1.6.6 (*Altered Nutrient Loading)* in Section 7.1 (*Dams).*

7.3.6.5 Altered Salinity

Dikes and levees also cause shifts in salinity within tidal marshes, caused by the alteration of flow regime and changes in tidal flushing. The impacts on HCP species from salinity changes are described in Section 7.6.6.6 (*Altered Salinity*) in Section 7.6 (*Tide Gates*).

7.4 Outfalls

Outfalls occur in a variety of environments and are built to move water from one place to another. They are typically associated with industrial processes, municipal wastewater treatment, and stormwater infrastructure. They can be categorized as submerged or exposed structures. Submerged outfalls are most common in lakes and marine waters, often associated with municipal and industrial wastewater and stormwater discharges. Marine outfalls that emerge at intertidal elevations are typically also considered submerged outfalls, as they are submerged at least some of the time. Exposed outfalls are outfalls that typically occur in riverine environments. Submerged and exposed outfalls are often screened to prevent fish entering the outfall pipe and to prevent large debris from exiting the outfall. Impacts on HCP species from fish screens are covered in the Fish Screen white paper (Herrera 2007b).

The mechanisms of impact from outfalls are the same six general impact pathways as for dams, weirs, and dikes: construction and maintenance activities, hydraulic and geomorphic modifications, ecosystem fragmentations, riparian vegetation modifications, aquatic vegetation modifications, and water quality modifications. Because many of the submechanisms of impact for outfalls are the same as for dams, this section contains extensive cross referencing, and the complete suite of submechanisms are not presented as independent subheadings. Where differences between dams and outfalls exist, additional information is presented. In addition, outfalls can be either submerged under water or exposed above the water surface. Therefore, where applicable, these two types of outfalls are discussed separately, and the organizational structure of this section may not follow exactly the organization of previous sections.

7.4.1 Construction and Maintenance Activities

Construction activities from outfalls have similar impacts on fish and invertebrates as dams from the use of heavy equipment. For example, during construction, burial of outfall pipes requires excavation of sediment that is typically used to backfill the excavated project area (Williams and Thom 2001). Equipment use will result in disturbance to riparian and aquatic vegetation and potentially result in bank failure. The construction “footprint” is likely smaller for outfalls compared to a dam; therefore, increased sediments and vegetation removal should be smaller compared to previously described activities for dams. Pile driving will likely not occur, so noise from pile driving will be minimal compared to a dam. However, underwater noise would likely result from construction equipment use and materials placement, including rock that is used to armor the outfall and dissipate energy from the outfall effluent.

Impacts from construction and maintenance activities from outfalls are mainly the result of bank disturbance, increased sediments, any dewatering and channel work needed, and any dredging required. These impact mechanisms are described in Section 7.1.1.2 (*Bank/Channel/Shoreline Disturbance*), Section 7.1.1.4 (*Dewatering, Flow Bypass, and Fish Handling*), Section 7.1.1.6 (*Construction and Maintenance Dredging*), and 7.1.6.3 (*Altered Suspended Sediments and Turbidity*), all in Section 7.1 (*Dams*).

7.4.2 Hydraulic and Geomorphic Modifications

The hydraulic and geomorphic impacts of outfalls are diverse. Outfall design and effluent characteristics play an important role in the degree of impact on fish and invertebrates. Well-designed outfalls that discharge small flow rates of effluent with similar constituents (i.e., temperature, salinity, turbidity and density) as the receiving water do not have significant hydraulic and geomorphic impacts (see Section 7.4 [*Outfalls*] for details). Impacts associated with submerged and exposed outfalls are discussed separately below.

7.4.2.1 Submerged Outfalls

Geomorphic impacts of submerged outfalls are related primarily to the expression of outfall plumbing that occurs above grade. Outfalls that are elevated above the natural grade have the potential to interrupt the natural flow of sediment along the shoreline (Herrera 2006b). In this case, outfalls act as groins, which have a number of indirect impacts on fish and invertebrates (as described below). Given the sensitive nature of the sediment supply along the shorelines of Puget Sound (Finlayson 2006), the interruption of longshore transport has the potential to be significant if it spans the beach foreshore, the zone of maximum sediment transport (Finlayson 2006), and so long as interruption occurs above the closure depth. For a detailed treatment of closure depth, see the *Breakwaters* section of the Shoreline Modifications white paper (Herrera 2007b) and Komar (1998). Only if the outfall protrudes above grade and below closure depth will the effects be minimal.

Hydraulic impacts are related primarily to the flow rate and the constituents emanating from the outfall. Typically, submerged outfall outlets are located below the closure depth and below significant light penetration, such that aquatic vegetation and fish use are limited. In these situations, hydraulic modifications likely have a minimal effect on fish and invertebrates; unless they modify basin-scale water-column constituents (see Section 7.4.6 [*Water Quality Modifications*]). However, if the effluent is of a different density than the ambient water, stratification of the basin can occur, which can have severe water quality impacts, most notably through eutrophication and benthic anoxia (Fischer et al. 1979). Also, to prevent the deposition of debris in the outfall and the diffuser ports, minimum velocities are often required (Fischer et al. 1979). Large velocities can alter nearshore circulation patterns, even if outfalls are sited in deep waters by mixing otherwise distinct water masses (Fischer et al. 1979). Scour can also occur as a result of large discharge velocities (Rice and Kadavy 1994).

If the outfall outlet is located above the closure depth, significant impacts on local geomorphology can occur. These may take the form of changing substrate, changing nearshore circulation patterns, and possibly excluding fish from key habitats with high velocities. These high velocities (or changes in nearshore circulation produced by them) could also remove aquatic vegetation. Because many of the HCP species use surface waters preferentially to deeper water, the impact on fish and invertebrates would be greater the shallower the outfall outlet is designed. However, the precise distribution of velocities and their change from preconstruction conditions would need to be determined with a hydraulic numerical model.

7.4.2.1.1 Altered Flow Regime–Altered Wave Energy

If outfalls or outfall pipes protrude above grade, alterations in local wave energy can occur. As hard points along the shoreline, outfall structures can result in the retention of wave energy in the surrounding area (Komar 1998). Regardless of the nature of the alterations, the modified relationship between topography and wave energy results in a shoreline that is out of equilibrium with natural shoreline processes (Komar 1998). Best management practices (BMPs) can generally reduce or eliminate these effects (see Section 11 [*Habitat Protection, Conservation, Mitigation, and Management Strategies*] for details). As a result, wave energy artificially accumulates in some areas and is diminished in others. This redistribution of wave energy can have a number of interrelated indirect and direct effects on fish and invertebrates, as described below in two categories: those that result in changes in substrate, and those that change water column characteristics.

Substrate Stressors

Substrate is an important factor controlling the growth of aquatic vegetation in Puget Sound (Koch 2001) and conditions for fish spawning in intertidal waters (typically forage fish). Increased wave energy intensifies the bed shear stress; if some of the coarsest material is not mobilized, a generally coarser substrate results (Komar 1998). However, the degree to which this occurs depends on the geologic setting. On the outer coast, for example, substrate is loose, deep, sandy, and unconsolidated. In protected, previously glaciated areas, the basin topography is complex and the coarse nature of the substrate slows down erosion dramatically (Nordstrom 1992). In these locales, a lag deposit can result in a near bedrock-like shoreline [e.g., Foulweather Beach (Finlayson 2006)]. Typically, however, hardening of the shoreline bed (beach substrate) manifests by a loss of the pebble veneer that is common throughout much of Puget Sound [(Finlayson 2006)]. This process is similar to what has occurred on the urbanized shorelines throughout the Great Lakes (Chrzastowski and Thompson 1994). These changes have produced pronounced ecological changes within recent years in the Great Lakes (Meadows et al. 2005), causing the elimination of native species and enabling invasives (zebra mussels) to more easily dominate the nearshore ecosystem (Marsden and Chotkowski 2001).

Changing substrate can adversely affect the growth of aquatic vegetation. For instance, eelgrass is incapable of growing in predominantly gravel substrates (Koch 2001). For details regarding the effects on fish and invertebrates as a result of marine aquatic vegetation disturbances, see Section 7.6.5 (*Aquatic Vegetation Modifications*) in Section 7.6 (*Tide Gates*).

Although there are few experimental studies of forage fishes, damage to surf smelt spawning areas has been documented in the presence of shoreline hardening (i.e., bulkheads) in Hood Canal (Herrera 2005; Penttila 1978; Thom et al. 1994). Typical spawning substrates for forage fishes consist of fine gravel and coarse sand, characteristic of the pebble veneer found throughout Puget Sound (Finlayson 2006), with broken shells intermixed in some cases (Thom et al. 1994). Surf smelt make no attempt to bury their demersal, adhesive eggs but rely on wave action to cover the eggs with a fine layer of substrate (Thom et al. 1994). Therefore, changing the wave environment may change the survivability of surf smelt spawn. The importance of substrate to spawning has also been empirically demonstrated in the closely related Japanese surf smelt (Hirose and Kawaguchi 1998).

Pacific sand lance spawn in the high intertidal zone on substrates varying from sand to sandy gravel. Sand lance also rely on sandy substrates for burrowing at night. As with surf smelt, sand lance spawning is susceptible to the deleterious effects of littoral alterations because sand lance rely on a specific beach profile with specific substrate compositions (Penttila 1995).

Deposition of large amounts of fine sediment can kill aquatic vegetation vital to nearshore HCP species. Recent work has shown that burying eelgrass to depths of as little as 25 percent of the total plant height could decrease productivity and increase the mortality of eelgrass (Mills and Fonseca 2003). Eelgrass can also be discouraged from colonizing new areas with high clay content as a result of recent sediment deposition (Koch 2001). For details regarding the effects on fish and invertebrates as a result of marine aquatic vegetation disturbances, see Section 7.6.5 (*Aquatic Vegetation Modifications*) in Section 7.6 (*Tide Gates*).

Water Column Stressors

Wave energy is the dominant source of fluid mechanical energy in the nearshore in most Washington waters (Finlayson 2006). Waves are responsible for mixing the upper portion of the water column (Babanin 2006) and producing high shear stresses near the bed (Lamb et al. 2004). These motions can prove harmful to aquatic vegetation and the fish and invertebrates that use and consume it.

Direct and Indirect Effects

Fish that are planktonic breeders have been shown to produce spatially variable spawn that relies on the combination of wave motion and ambient currents to be transported to appropriate and productive nursery areas (Hernandez-Miranda et al. 2003; Rooper et al. 2006). Waves produce water motion and induce transport both in the water column and near the seabed that are capable of transporting particulates large distances (Liang et al. 2007; McCool and Parsons 2004). Altering these mechanical processes alters transport rates (Liang et al. 2007; McCool and Parsons 2004).

While no specific studies have analyzed this effect for any of the HCP species, several inferences can be drawn based on their life history. Herring, sand lance, surf smelt, the rockfish, pollock, Pacific cod, hake, lingcod, Olympia oyster, and northern abalone all have planktonic eggs or larvae that are dependent on wave, current, and circulation patterns for transport to and/or retention in areas favorable for rearing. A broad array of research has demonstrated that the developing eggs or larvae of species with planktonic life-history stages that are transported into areas unfavorable for rearing face a high likelihood of starvation and predation or, in the case of schooling pelagic species, may be permanently isolated from their spawning population (Sinclair 1992). Therefore, it is possible that changes in wave-induced water movement could transport spawn and larvae to less-desirable areas and therefore contribute to mortality of the larvae of planktonic-breeding species.

Invertebrates that cannot tolerate extremely high shear stresses or burial may be directly affected by altered wave energy. While experimental evidence of the mortality limits of large shear stresses on mollusks or other invertebrates is lacking, Olympia oysters have been shown to be intolerant of siltation and do best in the absence of fine-grained materials (WDNR 2006b). The burial of mollusks and the related stress or mortality resulting from partial or complete burial have been addressed empirically (Hinchey et al. 2006). Results of these studies indicate that species-specific responses vary as a function of motility, living position, and inferred physiological tolerance of anoxic conditions. Most shorelines in Washington do not have the sedimentation rates necessary to bury mollusks. However, near major river mouths, sedimentation rates [e.g., on the Skagit as inferred by (Hood 2006)] are possible that exceed the criteria for mortality set forth by Hinchey et al. (2006).

Attenuation of waves can increase water column stratification in marine waters and lead to dissolved oxygen reduction and temperature anomalies (Qiao et al. 2006); see Section 7.1.6 (*Water Quality Modifications*) in Section 7.1 (*Dams*) for details. Surficial mixing and circulation also play an important role in primary productivity, particularly near large river mouths [e.g., Willapa Bay (Roegner et al. 2002)]. Disruption of these processes may have effects on the primary productivity and ultimately on any marine species through food-web interactions.

Wave energy also plays a role in the distribution of aquatic vegetation used by salmonids and other nearshore fishes, particularly in energetic environments. High wave energy has shown to inhibit the colonization and growth of some seagrasses [e.g., eelgrass (Fonseca and Bell 1998); see Section 7.6.5 (*Aquatic Vegetation Modifications*) under Section 7.6 (*Tide Gates*) for details], although in more recent work in Puget Sound, no correlation was found between eelgrass prevalence and wave characteristics (Finlayson 2006). High shear stresses associated with waves can also dislodge kelp (Kawamata 2001). Overall, altered wave energy can adversely affect HCP species through the movement of spawn and larvae to less desirable habitats, increased shear stress and burial, alteration of water column stratification, and the distribution of aquatic vegetation.

7.4.2.1.2 Altered Flow Regime–Altered Current Velocities

Submerged outfalls produce areas of increased current velocity. These strong currents can have significant impacts on both aquatic vegetation and the substrate in which it is embedded. The relationship between flow velocity and a change in substrate is related to that quantity known as the boundary shear stress (Miller et al. 1977). Substrate and aquatic vegetation are removed if a critical shear stress is exceeded. These impacts are comparable to those discussed in the preceding subsection (Section 7.4.2.1.1, *Altered Flow Regime-Altered Wave Energy*).

The alterations in local circulation can also reduce velocities elsewhere. These reductions could encourage the deposition of fine sediment (i.e., silt and clay) (Miller et al. 1977), particularly near sources of fine sediment [i.e., large rivers (Downing 1983)]. These effects are similar to those discussed in Section 7.4.2.1.1 (*Altered Flow Regime–Altered Wave Energy*).

Direct and Indirect Effects

As described for the effects associated with alterations in wave energy, alterations in velocity in the water column could alter transport and increase the mortality of planktonic spawn (e.g., Pacific herring). See the preceding submechanism (Section 7.4.2.1.1 [*Altered Flow Regime–Altered Wave Energy*]) for details.

Invertebrates cannot tolerate extremely high shear stress or burial. Experimental evidence of the mortality limits of large shear stresses on mollusks or other invertebrates is not available. Burial of mollusks is discussed in the preceding subsection (7.4.2.1.1 [*Altered Flow Regime–Altered Wave Energy*]).

Nearshore currents, even those in heavily altered environments, do not exceed the threshold for adult salmonid navigation, but high velocities have been shown to exclude some small fishes (e.g., juvenile salmonids and forage fishes) from navigating nearshore waters (Michny and Deibel 1986; Schaffter et al. 1983). This exclusion could cause the fragmentation of habitat by excluding fish from key areas or disrupting littoral migration for these species.

Outfalls have the potential to heighten water column stratification. Stratification can reduce flow velocity, mixing, and circulation (Fischer et al. 1979). Reduction in mixing can lead to eutrophication (see Section 7.4.6, *Water Quality*), and changes in circulation can have deleterious impacts on pelagic spawners (e.g., herring). Eelgrass and many other species of aquatic vegetation (e.g., bull kelp) also require some water motion for survival (Fonseca et al. 1983). These species could be impacted directly and other HCP species could be affected indirectly (see Section 7.4.5, *Aquatic Vegetation Modifications* for details).

7.4.2.1.3 Altered Flow Regime–Altered Nearshore Circulation

Nearshore circulation is a general phrase that describes the flux of salt, water, and sediment in association with tidal and wave motion near the shoreline. In more exposed, sandy settings, nearshore circulation is dominated by the mechanics of wave breaking (Komar 1998). These effects are generally insignificant in Puget Sound (Finlayson 2006), but they can be an important process when swell is present [i.e., on the outer coast (Komar 1998)]. In Puget Sound and near the mouth of large rivers (e.g., the Columbia), tidal currents and freshwater input play a more important role in nearshore currents. Outfalls can disrupt nearshore circulation and tidal flow. As such, they can disrupt transport of planktonic spawn and produce geomorphic changes that could be inhospitable to HCP species (e.g., the accumulation of silt and clay in forage fish spawning areas).

Direct and Indirect Effects

Like wave energy, nearshore circulation patterns are a dominant characteristic that shapes the suitability of nearshore habitats for a range of HCP species. Alteration of nearshore circulation patterns can produce many of the same effects described for altered wave energy in Section 7.4.2.1.1 (*Altered Flow Regime–Altered Wave Energy*). Specifically, fish and invertebrate species that are planktonic breeders have been shown to produce spatially variable spawn that relies on the combination of wave motion, ambient currents, and circulation patterns for transport to and retention in productive nursery areas (Hernandez-Miranda et al. 2003; Rooper et al. 2006; Sinclair 1992). As stated, while specific studies on HCP species are lacking, virtually all of the purely marine HCP species have a planktonic egg and/or larval life-history stage dependent on rearing habitat transport and retention dynamics. Developing eggs or larvae that are transported into areas unfavorable for rearing face a high likelihood of starvation and predation or, in the case of schooling pelagic species, may be permanently isolated from their spawning population (Sinclair 1992).

7.4.2.1.4 Altered Sediment Transport

If submerged outfall pipes or other related immobile infrastructure protrude above grade in nearshore settings, they have the potential to obstruct natural littoral transport (Herrera 2006b). Alteration of sediment transport patterns can present potential barriers to the natural processes that build spits and beaches and provide substrates required for plant propagation, fish and shellfish settlement and rearing, and forage fish spawning (Haas et al. 2002; Penttila 2000; Thom and Shreffler 1996; Thom et al. 1994). These impacts are primarily manifest as a change in substrate, the impacts of which are discussed in Section 7.4.2.1.1 (*Altered Flow Regime–Altered Wave Energy*).

Direct and Indirect Effects

The primary indirect effect of changing sediment supply is to alter the distribution of substrate within the littoral cell within which the modification occurs (Terich 1987). Therefore, the loss of sediment to a drift cell results in a coarsening of the substrate as fine-grained sediment is lost to deep portions of the basin by resuspension (Finlayson 2006), and this sediment is not resupplied by freshly eroded bluff sediments. The coarsening would have the same effects on fish and invertebrates as discussed in Section 7.4.2.1.1 (*Altered Flow Regime–Altered Wave Energy*). However, because some drift cells can be extremely long [e.g., more than 20 miles long in the drift cell that extends between Seattle and Mukilteo on the northeastern shore of the main basin of Puget Sound (Terich 1987)], the effects of a modification can extend well beyond the primary activity area.

The primary direct effect of an altered sediment supply on fish and invertebrates is to alter the turbidity in the nearshore environment (Bash et al. 2001; Berry et al. 2003). (See Section 7.1.6.3 [*Altered Suspended Solids and Turbidity*] in Section 7.1 [*Dams*] for a full discussion of the effects of turbidity.)

7.4.2.1.5 Altered Substrate Composition

Submerged outfalls require the installation of hard, immobile substrate (i.e., outfall plumbing is necessarily rigid by design). All piping associated with an outlet that is constructed above grade has the potential to enhance wave energy and initiate the colonization of invasive species (Marsden and Chotkowski 2001; Wasson et al. 2005). Often, riprap is placed around piping (e.g., scour pool) to protect it from nearshore erosion. The nutrient loading and exotic seed loading associated with many outfall effluents could exacerbate this problem (see Section 7.4.6 [*Water Quality Modifications]* for details).

Direct and Indirect Effects

The primary indirect effect of altered substrate composition on nearshore ecology is to encourage a shift toward hard-substrate, often invasive, communities (Wasson et al. 2005). However, the immobile substrate also fundamentally changes the mechanics of water motion on the shoreline, increasing wave reflection (Finlayson 2006; Komar 1998) and oftentimes altering the exchange of water into and out of the shoreline area if impermeable materials are used (Nakayama et al. 2007). These effects are discussed in preceding subsections (i.e., Section 7.4.2.1.1 [*Altered Flow Regime–Altered Wave Energy*] and Section 7.4.2.1.3 [*Altered Flow Regime–Altered Nearshore Circulation*]).

In general, the addition of immobile substrate decreases habitat suitability for juvenile salmonids and changes the character of the shoreline that was previously conducive to their use [e.g., (Knudsen and Dilley 1987; Li et al. 1984; Peters et al. 1998; Schaffter et al. 1983)]. While data indicate that the habitat utilization of riprapped banks by yearling and older trout species may be equal to or higher than natural banks, use by subyearling trout, coho, and Chinook salmon is lower (Beamer and Henderson 1998; Garland et al. 2002; Hayman et al. 1996; Knudsen and Dilley 1987; Schmetterling et al. 2001; Weitkamp and Schadt 1982). Knudsen and Dilley (1987) found that the abundance of juvenile salmonids was reduced by bank reinforcement activities due to a loss of structural diversity and that these reductions were correlated with the severity of habitat alteration, the size of the stream, and the size of the fish. Size of material is also relevant, as greater fish densities have been generally correlated with larger rock (Beamer and Henderson 1998; Garland et al. 2002; Lister et al. 1995). Lister et al. (1995) found that salmonid densities were greater along banks with riprap greater than a 1-ft (30-cm) median diameter compared to natural banks composed of cobble-boulder material. In Elliott Bay, Toft et al. (2004) found similar densities of juvenile salmonids at sand/cobble beaches and riprap sites in settings where the riprap extended only into the upper intertidal zone. When riprap extended to the subtidal zone, higher densities of juvenile salmonids were found along riprap than at sand/cobble beaches. Toft et al. (2004) hypothesized that this finding may be based on the fact that the shallow-water habitats preferred by juvenile salmonids were compressed along the highly modified shorelines with steep slopes; therefore, their snorkel observations were able to record all juvenile salmonids present. In comparison, at the sand/cobble beaches, the slopes were gentler, the zone of shallow water was much wider, and densities were therefore lower because the fish were more dispersed.

It is possible that coarser substrate could benefit some HCP species. An active debate in the scientific community is whether shoreline hardening structures are as productive and diverse as natural hard-rock shorelines, particularly in the Adriatic Sea east of Italy (Bacchiocchi and Airoldi 2003; Bulleri and Chapman 2004; Guidetti, Verginella et al. 2005). In addition to the elimination of shifting, sandy habitats, the Adriatic Sea studies have shown that maritime structures caused increases in piscivores and urchins, as well as decreased numbers of native species that prefer more mobile substrates (Guidetti, Bussotti et al. 2005). Although species distributions are clearly different in Italy than in Washington State, the steep paraglacial landscape, relatively short period, and locally generated waves make the hydraulic and geomorphic variables essentially identical (Finlayson 2006). Complicating the debate are other maritime activities that affect fish and invertebrates (i.e., fishing and ship traffic), which are difficult to separate from shoreline hardening and likely limit any gain in the transition of habitat type (Blaber et al. 2000; Guidetti, Bussotti et al. 2005).

7.4.2.2 Exposed Outfalls

The most important hydraulic and geomorphic effect associated with exposed outfalls is the ability for the outfall to create a scour pool at its outlet. Often in the outfall design, riprap or other immobile surfaces are added to prevent erosion at the outlet. This protective material often protrudes into the channel or floodplain. These modifications potentially will have a significant impact on the substrate surrounding the outfall outlet, causing additional scour. Scour can affect downstream habitats by transporting and depositing fine sediments, thereby compromising downstream spawning habitat. It can also dramatically modify the types and abundance of substrates available to support aquatic vegetation that is important to a suite of HCP riverine species.

However, exposed outfalls can also protrude into a stream or river channel and intercept the flow of sediment downstream. In this case, the outfall behaves like a groin and can disrupt the substrate in the vicinity of these alterations. Because HCP species depend on the presence or absence of particular substrate types to support important life-history functions, changes in sediment source contributions can have direct and indirect effects on those species.

7.4.2.2.1 Altered Flow Regime

Protruding, exposed outfalls can alter the velocity field in riverine environments by redirecting flow away from the banks and toward the center of the channel, just as groins can do (Lagasse et al. 2001). The formation of flow-separation eddies adjacent to these structures results in areas of relatively low velocity in these areas and along the protected bank (Lagasse et al. 2001). The net effect is to confine the flow, contributing to increased velocity and bed scour. Protection of the outlet with riprap can also reduce the hydraulic roughness and can also increase velocity and bed scour. If outfalls do not protrude and their effluent exits at a small velocity, their impact on the flow regime is negligible.

Direct and Indirect Effects

Fish and invertebrates inhabiting riverine environments require certain flow velocities for spawning, rearing, and foraging. For example, increases in flow velocities could present potential barriers to fish migration or could exceed thresholds for certain life-history stages of some HCP species. For instance, leopard and Umatilla dace inhabit riverine environments where the velocities are less than 1.6 ft/sec (Wydoski and Whitney 2003). Exceeding this velocity as a result of outfall installation would render habitat unsuitable for these species. Flow increases may also increase scour and result in increased suspended sediment transport and alteration of channel depths. Effects on fish and invertebrates from increased suspended sediments are discussed in Section 7.1.6.3 (*Altered Suspended Solids and Turbidity*), and effects from changes in channel depths are discussed in Section 7.1.2.1.2 (*Altered Channel Geometry*) both in Section 7.1 (*Dams*).

7.4.2.2.2 Altered Sediment Transport

Stabilization measures, such as the placement of riprap near an outfall outlet, can reduce the supply of suitably sized substrates for spawning fish and invertebrates by limiting natural processes of channel migration and bank erosion. Protruding outfalls can reduce the local supply of coarse sediment by deflecting bed sediment from the riverbank to the center of the channel. Because the rate and caliber of sediment supplied to a channel can influence the substrate size (Dietrich et al. 1989), changes in sediment supply can alter the composition of substrate used by HCP species.

Direct and Indirect Effects

Fish and invertebrates require a range of substrate conditions in riverine environments for various life-history stages. Maintaining these conditions is dependent on the replenishment of suitably sized substrates to offset natural sediment transport processes that remove sediment. In a study in California, the primary cause for the decline of salmon in the Sacramento River was linked to the loss of spawning gravels normally derived from bank erosion before riprap bank stabilization (Buer et al. 1984).

7.4.2.2.3 Altered Substrate Composition

Outfalls can alter the composition of bed and bank materials by virtue of adding material coarser than the ambient bed or by adding flow and coarsening the existing sediments. Further, if the outfall extends into the channel, it can deflect high-velocity flows to the center of the channel and induce flow separation. Outfalls can also initiate the deposition of fine sediments leeward of the protruding structure.

Placement of outfalls above grade eliminates the potential to maintain riparian vegetation. This can increase the flow velocity and increase the potential for scour and substrate coarsening through a reduction in hydraulic roughness compared to vegetated conditions (Millar and Quick 1998).

Increased velocities associated with flow constrictions created by protruding outfalls can indirectly affect HCP species by causing local bed scour around structures and corresponding sediment deposition downstream (Richardson and Davis 2001). In addition, high-velocity effluent can initiate bed scour, causing the selective removal of finer sediment, coarsening the substrate. Likewise, deposition of the finer materials originating from the outfall downstream can bury organisms and cause the substrate to become finer.

Direct and Indirect Effects

Alteration of the substrate composition through coarsening or fining of the bed can have direct and indirect effects on HCP species. These impacts from substrate alterations are discussed in Section 7.1.2.1.4 (*Altered Substrate Composition*) in Section 7.1 (*Dams*).

7.4.2.2.4 Altered Groundwater–Surface Water Interactions

Most outfalls do not require the use of pilings or other impermeable structures that impede the exchange of hyporheic water with main river channels. However, if an outfall is placed parallel to a river or stream channel and it is sufficiently large, the outfall has the potential to disrupt or eliminate hyporheic exchange, reduce lateral habitat connectivity, and alter stream temperatures buffered by groundwater inputs. For a discussion of the impacts of lateral habitat disconnection, see Section 7.1.3.1.2 (*Altered River–Floodplain Connectivity*) under Section 7.1 (*Dams*).

Direct and Indirect Effects

Structures that alter groundwater dynamics in riverine systems can directly affect fish and invertebrates in the short term by influencing water quality and habitat suitability or availability. In the long term, changes to groundwater exchange can generate indirect effects on fish and invertebrate species by affecting low-flow conditions (i.e., increasing the magnitude of periods of drought resulting in reduced habitat availability and suitability, potential stranding, or desiccation), as well as by affecting water quality through warmer stream temperatures and decreased organic and nutrient inputs. For more details on the impacts on fish and invertebrates from the loss of groundwater–surface water interactions and subsequent water quality effects, refer to Section 7.1.2.1.5 (*Altered Groundwater–Surface Water Interaction*), Section 7.1.6.1 (*Altered Temperature Regime*), Section 7.1.6.6 (*Altered Nutrient Loading*), all in Section 7.1 (*Dams*).

7.4.3 Ecosystem Fragmentation

Ecosystem fragmentation (as described previously in this white paper) would be minimally affected by outfalls. Habitat connectivity will likely remain intact for most outfalls, both submerged and exposed. If an outfall crosses a stream or river such that it interferes with downstream flow, then impacts related to upstream–downstream connectivity will be important. These impacts are described in Section 7.1.3.1.1 (*Altered Longitudinal Connectivity*) in Section 7.1 (*Dams*). If an outfall is placed along the bank of a stream or river, it could potentially exclude access to side channel and floodplain habitat. If so, species that use floodplain habitats would be affected. A discussion about the impacts on fish from lateral habitat loss is provided in Section 7.1.3.1.2 (*Altered River–Floodplain Connectivity*) in Section 7.1 (*Dams*). When exposed outfalls are located such that they terminate at the riverbank and are located above the stream, their main impact is the result of water chemistry changes, if the effluent contains toxic materials (see Section 7.4.6 [*Water Quality Modifications]* for details). However, some studies have shown that species diversity and composition changes can be minimal above and below an outfall (Fries and Bowles 2002; Pillard 1996). These studies suggest that ecosystem fragmentation with respect to altered species composition and diversity is likely minimal.

Marine outfalls may cause ecosystem fragmentation as a result of altered wave energy, current velocities, and nearshore circulation. The degree of these impacts depends on the volume of discharge and local mixing and may result in impacts on some HCP species. These impacts are discussed in Section 7.4.2.1 (*Submerged Outfalls*).

7.4.4 Riparian Vegetation Modifications

Modification of riparian vegetation in riverine and marine environments from the construction or maintenance of an outfall will generally be minimal. However, removal of riparian vegetation during construction and maintenance is possible. In addition, if the outfall is causing significant downstream erosion due to large changes in flow, bank failure may result in a loss of riparian vegetation. If removal of riparian vegetation is significant, HCP species would be exposed to stressors caused by a variety of impact mechanisms:

- Altered shading, solar input, and ambient air temperature
- Altered bank and shoreline stability
- Altered allochthonous inputs
- Altered groundwater–surface water interactions
- Altered habitat complexity
- Increased nutrient/pollutant loading

These submechanisms of impact and related ecological stressors are not subactivity specific and are described in detail in Section 7.1.4 (*Riparian Vegetation Modifications*) in Section 7.1 (*Dams*).

Riparian areas in marine environments serve similar functions to those in riverine environments. Therefore, modification of marine riparian vegetation associated with outfall construction will likely result in adverse impacts on several HCP species. However, these impacts will likely be minimal. A detailed discussion of the impacts from modification of marine riparian vegetation on HCP species is provided in Section 7.6.4 (*Riparian Vegetation Modifications*) in Section 7.6 (*Tide Gates*).

7.4.5 Aquatic Vegetation Modifications

Impacts on aquatic vegetation will likely occur in proximity to the outfall structure and would not pose any large-scale effects on HCP species. However, increased flows may cause scour and loss of aquatic vegetation. In addition, construction and maintenance activities may bury aquatic vegetation.

Any activity that mechanically removes or by other means affects aquatic vegetation will reduce the sediment, nutrient, and pollutant retention and reduction capabilities of the system. Indirect impacts from removal of aquatic vegetation will increase nutrient and pollutant loading to receiving waters, which could exacerbate eutrophic conditions and/or metals toxicity. A detailed discussion of the impact on species from nutrient loading is presented in Section 7.1.6.6 (*Altered Nutrient Loading)* in Section 7.1 *(Dams)*.

Increased nutrient loading from outfall effluent may result in an indirect effect on aquatic vegetation. Increased nutrient loading can stimulate primary productivity and lead to decreased dissolved oxygen. Eutrophication can lower dissolved oxygen concentrations and negatively affect HCP species. Impacts from lowered dissolved oxygen are described in detail in Section 7.1.6.2 (*Altered Dissolved Oxygen*) in Section 7.1 (*Dams*).

7.4.6 Water Quality Modifications

The most significant impacts on HCP species from outfalls are related to water quality modifications as a result of the presence of pollutants in the discharged effluent. Urban runoff, wastewater treatment plant effluent, and combined sewer overflows are the source of nutrients, sediment, metals, PAHs, and pesticides, all of which can change the chemistry and temperature of the receiving waters (Barber et al. 2006; Grapentine et al. 2004; Mulliss et al. 1997; Wenning et al. 1999). For example, outfalls from fish hatcheries can increase suspended sediments in the receiving waters (Fries and Bowles 2002). Increased temperatures from a power plant outfall can affect migration patterns of stingrays at Seal Beach (California) (Vaudo and Lowe 2006), while increased nutrients in sewage outfalls can contribute to increases in the productivity of the receiving water (deBruyn et al. 2003). In Canada, hormonally active chemicals have been shown to accumulate in the local white sucker, disrupting reproductive activities in females (Hewitt et al. 2005). In addition, stormwater flows can increase flows and erosion far downstream of the outfall structure (Williams and Thom 2001), causing increased turbidity. If erosion is severe enough, it could result in bank failure and landslides, further affecting the stream with increased sediment loading (Williams and Thom 2001).

Impacts on aquatic invertebrates from outfalls have also been documented. In the Clinch River (Virginia), effluent from a wastewater treatment plant that contained monochloramine and unionized ammonia from domestic effluent resulted in a 2.3-mile (3.7-km) reach below the outfall devoid of several freshwater mussels (*Unionidae*) (Goudreau et al. 1993).

Impacts on fish and invertebrates from altered temperatures, dissolved oxygen, turbidity, contaminated suspended sediments, pH levels, nutrient loading, and toxic substances are described in detail in Section 7.1.6 (*Water Quality Modifications*) in Section 7.1 (*Dams*). Because metals are commonly found in outfall discharges and are toxic to many aquatic organisms, a discussion of metal toxicity is presented below.

7.4.6.1 Metal Toxicity

In urban environments, metals loading to local waterways and water bodies from anthropogenic sources is a major pathway for aquatic habitat degradation. The primary metals of concern in the surface waters of Washington are copper, zinc, arsenic, lead, and nickel (Embrey and Moran 2006). Metals above threshold concentrations act as carcinogens, mutagens, and teratogens in fish and invertebrates (Wohl 2004). Additionally, the sublethal effects of copper toxicity have been extensively studied, with reported effects including impaired predator avoidance and homing behavior (Baldwin et al. 2003). The Washington State Department of Ecology has established water quality standards for marine and fresh waters for each of these constituents. These standards, issued in WAC 173-201A, are listed in Table 7-6. Freshwater toxicity thresholds are hardness-dependent and can vary widely depending on calcium and magnesium carbonate concentrations. The standards presented here are based on median hardness concentrations estimated from an extensive 3-year data set (2001–2003) from the Green River watershed (Herrera 2007c).

Table 7-6. Water quality criteria for metals in marine and fresh waters of the state of Washington.

Constituent	Freshwater		Marine	
	Acute	Chronic	Acute	Chronic
Arsenic	360	190	69	36
Copper	7	7.5	4.8	3.1
Lead	22.9	1.5	210	8.1
Nickel	640	104	74	8.2
Zinc	51.6	69.2	90	81

Units: ports per billion (ppb).
Adapted from: WAC 173-201A.

7.4.6.1.1 Direct and Indirect Effects

Metals are widely known to adversely affect fish species. Increased levels of copper and cadmium have been shown to cause mortality and lower growth rates in bull trout (Hansen, Welsh, Lipton and Cacela 2002; Hansen, Welsh, Lipton and Suedkamp 2002). Some species are more tolerant than others; for example, bull trout are more tolerant of zinc and copper compared to rainbow trout in laboratory studies (Hansen, Welsh, Lipton, Cacela et al. 2002). Recent studies have shown that even low concentrations of some metals can cause behavioral effects in juvenile salmonids. Dissolved copper, even at low concentrations, is a neurotoxin and damages the sensory capabilities of juvenile salmonids (Hecht et al. 2007). These effects can manifest over a period of minutes or hours and persist for weeks. In addition, copper can affect avoidance behavior; benchmarks developed by NOAA Fisheries showed that a range of 0.18–2.1 parts per billion (ppb) dissolved copper above background levels (for ambient waters below 3 ppb) were a cause for concern (Hecht et al. 2007).

The effects of increased metals on invertebrates are important because many metals adsorb onto sediment particles. Those invertebrates that reside in sediment and filter feed (e.g., California floater, Olympia oyster) are more susceptible to increased metal loading and biomagnification in tissues.

7.5 Intakes and Diversions

Water diversion systems (including intakes) are built for a variety of reasons including, but not limited to: irrigation, hatcheries, power plants (hydropower, fossil fuel, and nuclear), water supply, general manufacturing, timber processing, and other purposes (e.g., creation of fish habitat). Most diversion systems route water through a concrete channel and/or enclosed pipe. Figure 7-2 illustrates general schematics of the most common forms of water diversion and intake systems. Diversion systems built in freshwater environments can either work by gravity or water pump, with gravity systems employed predominantly in riverine systems that provide the necessary head loss. For both, the diversion channel or pipe may run parallel or away from the stream channel.

Gravity fed diversions usually include a dam or weir-like structure that partially or fully spans a river or stream channel. The flow control structure is used to create the hydraulic head necessary to divert the water out of the channel (Figure 7-2A). Construction of a gravity-fed diversion system includes the design and installation of flow control dam or weir and a diversion channel. This diversion channel is typically made of concrete but can also consist of a metal pipe. This type of diversion system typically includes a fish screen located at the downstream end of the diversion channel.

Pumped diversion systems typically take the form of a pump house and intake pipe or gallery with an associated concrete channel or a pipe used to transport the diverted water to its intended use (Figure 7-2B). This type of system is commonly located along the bank of a stream/river or lake. This type of system is used where water must be pumped up and out of the source body because the necessary hydraulic head for a gravity diversion is not available. This type of diversion system typically includes a fish screen at the pump intake.

Gravity-fed and pumped diversion systems can also be combined by having a pump station located at the end of the diversion channel. In these cases, fish screens are typically located at the pump intake.

In marine and lacustrine environments, water intake systems typically consist of an intake pipe with an associated pumping system. The intake may extend some distance into the water body while the pumping system is located onshore (Figure 7-2C). This type of intake system typically includes a fish screen located at the pipe's mouth, although some configurations may incorporate additional internal screening mechanisms. Intake systems fed by tidal exchange may also be employed in certain settings. This type of configuration may incorporate a tide gate on the shoreline to regulate intake flows. Construction of intake systems includes the design and installation of a pipeline and pump house and, potentially, a shoreline tide gate in marine environments. As with outfalls, the pipe associated with these diversion systems can be categorized as submerged intakes (typical in reservoirs, lakes, and marine environments), and exposed intakes (which are found in stream and river environments). As is typically the case with outfalls, intakes are usually screened.

The impact mechanisms arising from intakes and diversions are the same six general impact pathways as apply to dams, weirs, dikes and levees, and outfalls: construction and maintenance activities, hydraulic and geomorphic modifications, ecosystem fragmentation, riparian vegetation modifications, aquatic vegetation modifications, and water quality modifications. Because many of the submechanisms of impacts for intakes and diversions are the same as those for other subactivity types, this section contains extensive cross referencing and the complete suite of submechanisms are not presented as independent subsections. Where available, information that is specific to intakes and diversions is presented as appropriate.

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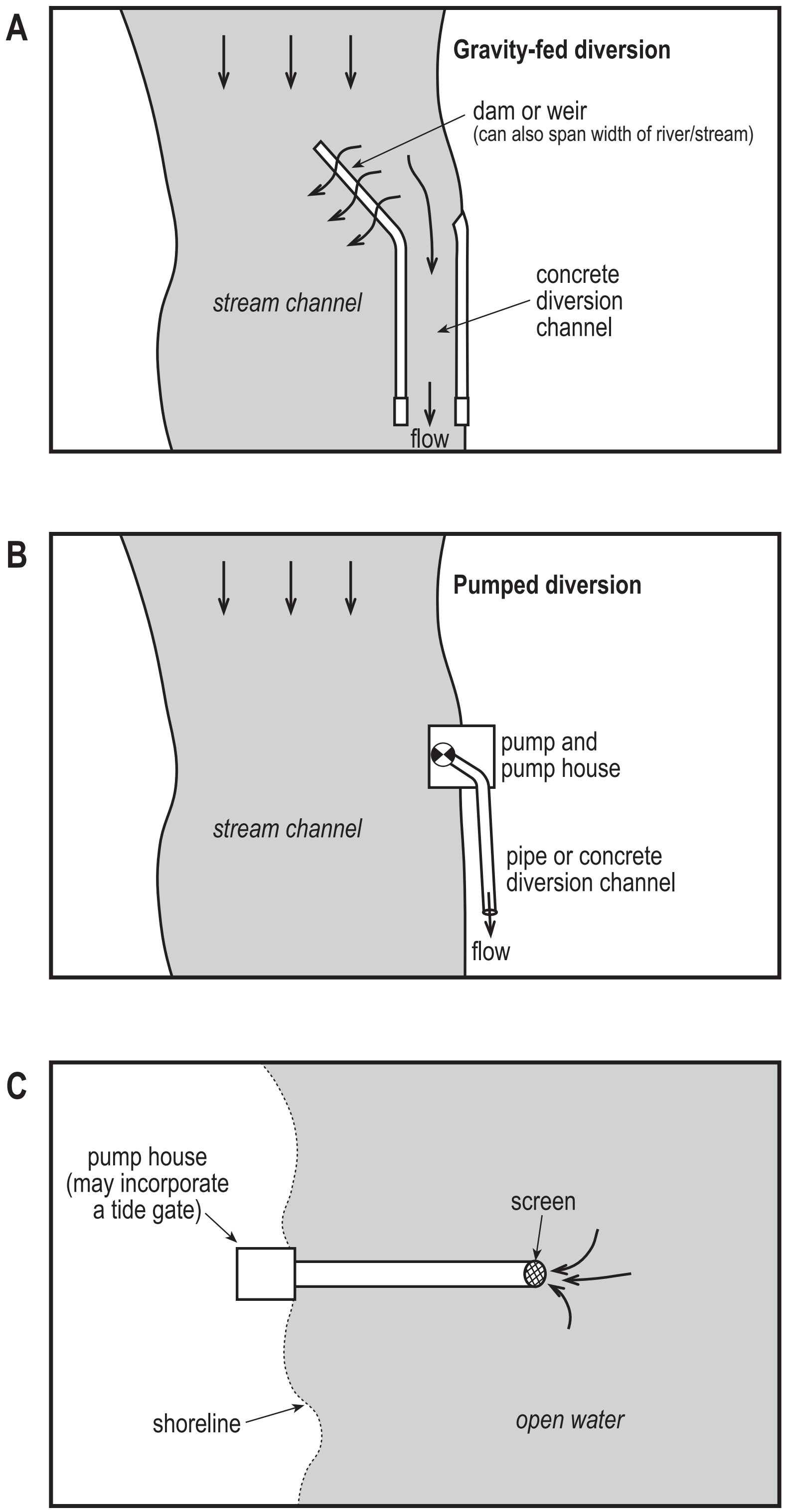

Note: In riverine environments, diversion systems can be gravity fed (A) or pumped (B). In marine environments, diversion systems are commonly constructed on the shore and pipelines extend out into the open water (C). In lacustrine environments, several configurations are possible including those shown in B and C.

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Figure 7-2. Types of diversion systems.

7.5.1 Construction and Maintenance Activities

Construction activities for intakes and diversions will have impacts on fish and invertebrates similar to those for dams, although the impacts will likely be of a smaller magnitude, depending on the structure. Activities associated with construction and repair of intakes and diversions pose the risk of increasing suspended solids, removing or disturbing aquatic and riparian vegetation, disturbing banks and shorelines, and releasing toxic substances to fresh and marine waters from construction materials and/or construction equipment. Construction activities may also include filling and dredging that can entrain organisms or permanently remove habitat for burrowing and benthic animals.

Construction and maintenance of intakes and diversions will likely result in smaller impacts (depending on the structure) from increased noise compared to a weir or a dam, particularly if pile driving is not included. For details on construction and maintenance activities and effects on fish and invertebrates and their habitats, see Section 7.1.1 (*Construction and Maintenance Activities*) in Section 7.1 (*Dams*).

7.5.2 Hydraulic and Geomorphic Modifications

Intakes and diversions may involve a number of hydraulic and geomorphic modifications. As with outfalls, the design of pipes or diversion channels transporting water into upland infrastructure can interfere with the transport of sediment, if those pipes are exposed above grade. Typically, this results in coarsening and erosion of the substrate in the lee of the pipe, as well as deposition and fining on the upstream or updrift side of the pipe. These impacts can usually be avoided by constructing the pipe below grade. See Section 7.4.2.1 (*Submerged Outfalls*) for details with regards to these impacts.

Other hydraulic and geomorphic modifications are related to altered flow regimes and changes in channel geometry. For example, diversions reduce flows downstream, which can lead to habitat loss (Kingsford 2000) and changes in channel width, depth, and velocity (Dewson et al. 2007).

In addition, other modifications are unavoidable and unique to the intake structure. In particular, inflowing water can attract fish toward the intake structure. All intakes should be screened in some manner to exclude fish. A full discussion of the impacts of screens, including these alternative measures, on fish and invertebrates is presented in the Fish Screen white paper (Herrera 2007b).

7.5.2.1 Direct and Indirect Effects

For direct and indirect impacts associated with changes in flows and channel geometry, refer to Section 7.1.2.1.1 (*Altered Flow Regime*) and Section 7.1.2.1.2 (*Altered Channel Geometry*), in Section 7.1 (*Dams*).

Unscreened intakes represent a severe hazard to all fish and their larvae; entrainment by an unscreened intake can cause mortality to all life stages of fish that inhabit areas near intakes (Newbold and Iovanna 2007). The area of influence of an intake is highly site- and design-dependent (Edinger and Kolluru 2000). To identify the area of influence of the intake, flow near any proposed unscreened intake should be investigated with a suitable hydraulic model.

The primary effect on invertebrates would be related to displacement of natural substrates. The emplacement of hard surfaces, either from the intake itself or piping connecting it to upland infrastructure, presents a surface on which invasive species can colonize. In the Great Lakes, extensive colonization by zebra mussels has completely clogged intake pipes (Ram et al. 1992).

7.5.3 Ecosystem Fragmentation

The presence of intakes and diversions will adversely affect HCP species through direct fragmentation of ecosystems. Diversions and intakes may alter flow, which can influence habitat connectivity and lead to habitat loss. Depending on the size of the diversion, changes in flow may be minimal or significant. Reduced discharges from diversions will reduce floodplain connectivity for those fish and invertebrates using these habitats (Kingsford 2000). Reduction in flow has also been shown to concentrate macroinvertebrates as a result of the reduction in available habitat, leading to increases in insect densities in the system (Dewson et al. 2007). Lateral fragmentation and loss of instream habitat will lead to impacts on HCP species. These impacts are discussed in detail in Section 7.1.3.1.2 (*Altered River-Floodplain Connectivity*) in Section 7.1 (*Dams*); for a summary of the effects from flow reduction, see Section 7.1.2.1.1 (*Altered Flow Variability*) in Section 7.1 (*Dams*). Although not widely documented, flow reductions inherently alter hyporheic exchange and groundwater–surface water interactions. For impacts on fish and invertebrates from reduced hydrologic exchange, see Section 7.1.3.1.3 (*Altered Groundwater-Surface Water Interactions*) in Section 7.1 (*Dams*). Water diversions have also been shown to alter water quality, namely temperature, which has been shown to change macroinvertebrate communities (Miller et al. 2007). For impacts on fish and invertebrates from altered temperatures, see Section 7.1.6.1 (*Altered Temperature Regime*) in Section 7.1 (*Dams*).

In addition to direct effects on habitat, intakes alter food webs and predator-prey interactions. Intakes and diversion can remove important resources from the aquatic ecosystem when they are entrained in the water column. For example, predator-prey relationships are altered when drifting insects and larvae are entrained in intake waters, effectively removing food resources from downstream organisms. For example, Benstead et al. (1999) showed that entrainment of freshwater shrimps in Puerto Rico can vary from 34–62 percent of drifting larvae based on field data and a flow model using 30 years of discharge data. In Hawaii, McIntosh et al. (2002) studied the impacts of diversions on riffle macroinvertebrate communities. The authors collected larval populations upstream and downstream of diversions and showed that total density decreased by 54 percent, thereby affecting trophic interactions downstream. Any alteration of natural food webs could change the species composition which, in turn, may allow the invasion of exotic species. The impacts from changes in food webs are similar to those discussed previously for dams (see Section 7.1.3.1.5 [*Altered Community Composition*] under Section 7.1 [*Dams*]).

Finally, the loss of large woody debris resulting from intakes and diversions is expected to be minimal and is not likely to adversely affect HCP species. However, some larger diversion structures may limit the passage of large woody debris and would have impacts similar to those discussed in Section 7.1.3.1.3 (*Alteration of Groundwater–Surface Water Interactions*) under Section 7.1 (*Dams*).

7.5.4 Riparian Vegetation Modifications

During construction and maintenance activities associated with intakes and diversions, riparian vegetation modification may occur. Removal or disturbance of riparian vegetation during construction of activities permitted under HPAs can expose HCP species to stressors caused by a variety of impact mechanisms:

- Altered shading, solar input, and ambient air temperature
- Altered bank and shoreline stability
- Altered allochthonous inputs
- Altered groundwater–surface water interactions
- Altered habitat complexity
- Increased nutrient/pollutant loading.

These submechanisms of impact and related ecological stressors are not subactivity specific and are described in Section 7.1.4 (Riparian Vegetation Modifications) in Section 7.1 (Dams). However, due to the usually small size of intake and diversion projects, changes in riparian vegetation will likely be minimal. The degree of impacts from this subactivity should be assessed on a site-by-site basis.

7.5.5 Aquatic Vegetation Modifications

Construction and maintenance activities associated with intakes and diversions may lead to the burial or scour of aquatic vegetation. Any activity that mechanically removes or by other means affects aquatic vegetation will reduce the sediment, nutrient, and pollutant retention and reduction capabilities of the system. For example, flow reduction from water diversion in Australia resulted in reductions in aquatic vegetation and poor vegetative health (Kingsford 2000). Indirect impacts from the removal of aquatic vegetation may increase nutrient and pollutant loading to receiving waters, which could exacerbate eutrophic conditions and/or metals toxicity. A detailed discussion of the impact on species from nutrient loading is presented in Section 7.1.6.6 (*Altered Nutrient Loading)* in Section 7.1 *(Dams)*, and Section 7.4.6.1 *(Metal Toxicity)* in Section 7.4 *(Outfalls)*.

Impact mechanisms and related ecological stressors from changes in aquatic vegetation are not subactivity specific and are described under dams in Section 7.1.5 (*Aquatic Vegetation*

Modifications). However, due to the generally small size of intake and diversion projects, changes in aquatic vegetation will likely be minimal. The degree of impacts from this subactivity should be assessed on a site-by-site basis.

7.5.6 Water Quality Modifications

Water quality modifications arising from diversions and intakes are similar to those discussed previously in Section 7.1 (*Dams*) and do not represent additional subactivity-specific modifications. Changes in flow regime can adversely affect receiving water temperatures as a result of reduced flow velocities. Alteration of hyporheic exchange and groundwater interactions will affect both temperature and nutrient cycling. Changes in riparian vegetation will also affect temperature, primary productivity, and nutrient cycling. Increased nutrient loading from a reduction in nutrient uptake by riparian and aquatic vegetation alteration can cause downstream eutrophication and decreased dissolved oxygen levels. A detailed discussion of the impacts from changes in temperature, dissolved oxygen, and nutrient and pollutant loading is presented under dams in Section 7.1.6 (*Water Quality Modifications*).

7.6 Tide Gates

Tide gates (also referred to as flood gates) are typically built to control tidal or floodwater inundation in low-lying areas. These structures are typically integrated into dikes and levees and are commonly used to drain river deltas and estuarine lowlands for conversion to agricultural or industrial uses. They allow water to drain from low-lying areas to marine or estuarine receiving waters while preventing the backflow of tidal or floodwater. In agricultural areas, tide gates prohibit salt water from entering croplands. In addition, tide gates lower the water table, pushing the anoxic layer deeper in the soil and promoting crop growth. Tide gates are commonly located at the mouths of streams or rivers where the estuary begins, or where tidal nonriverine channels drain ditches, fields, marshes, and small tributaries (Figure 7-3).

Tide gates come in many forms—from simple culverts through an earthen dike, to complex concrete structures that include deflecting walls and pilings both upstream and downstream of the structure (Figure 7-4). Associated with these structures are tide or flood boxes that restrict flow in one direction. Tide boxes can be either top-hinged or side-hinged and, depending on the type of gate, it will be open for shorter or longer times. The amount of time a gate is open is a function of the design, size, and weight of the tide box. The magnitude of tidal or floodwater fluctuations also influences the time the gate remains open. Tide gates generally have negative ecological consequences; however, the effects of this subactivity type have only recently received attention, and the bulk of this research has focused on gates in estuarine systems (Giannico and Souder 2005).

The mechanisms of impact from tide gates are the same six general impact pathways employed throughout this white paper: construction and maintenance activities, hydraulic and geomorphic modifications, ecosystem fragmentation, riparian vegetation modifications, aquatic vegetation modifications, and water quality modifications. Many of the submechanisms and stressors imposed by tide gates are similar to those imposed by other subactivity types discussed in this

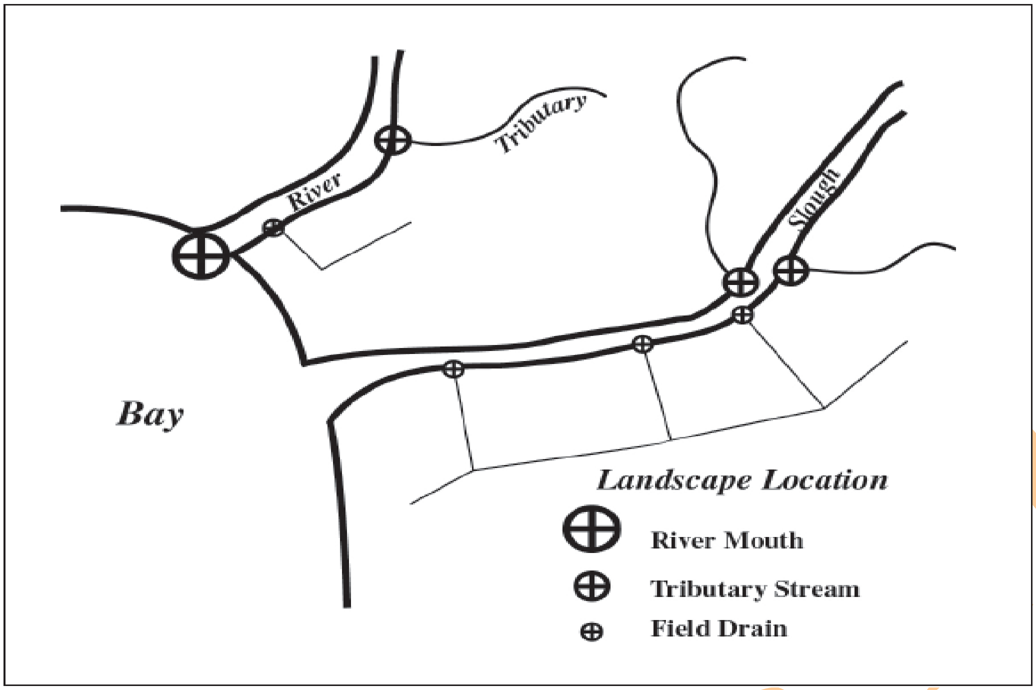

Figure 7-3. Common tide gate locations at the mouth of estuaries, tributary streams, and tidal nonriverine channels. Adopted from (Giannico and Souder 2005)

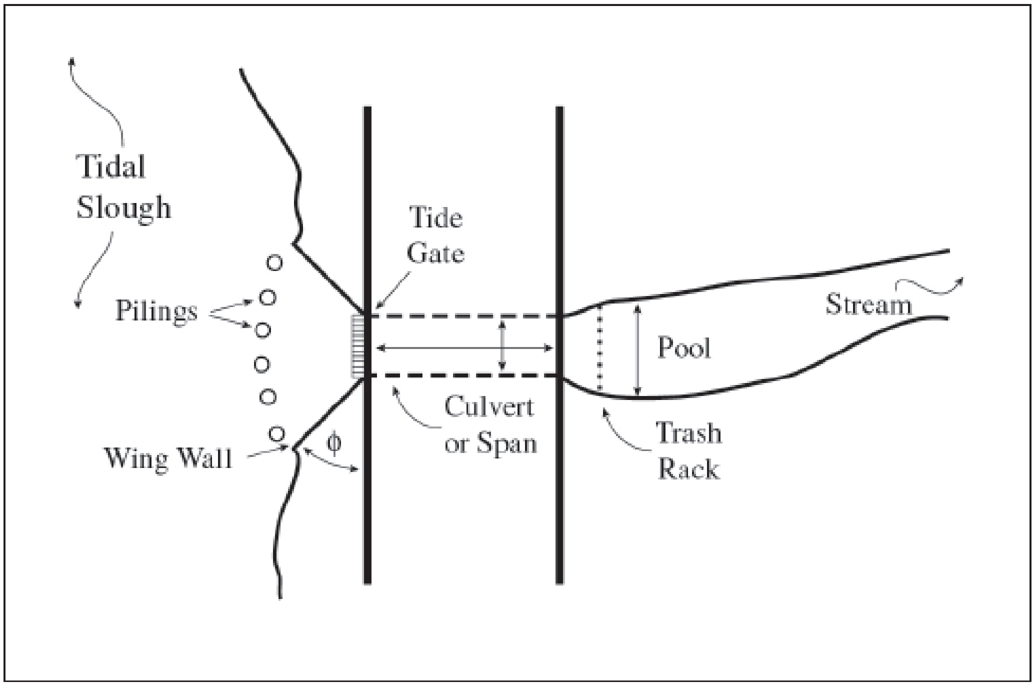

Figure 7-4. View of tide gate and supplemental features such as wing walls and pilings. Adopted from (Giannico and Souder 2005).

white paper; therefore, this section contains extensive cross-referencing where comparable effects have already been described, and the complete suite of submechanisms of impact are not presented as independent subsections. Where information specific to tide gates is available or relevant, it is presented below.

7.6.1 Construction and Maintenance Activities

There are many different types of tide gates, and impacts from construction and maintenance activities vary accord to the type of construction. However, these impacts are similar to those discussed previously for dams (Section 7.1). For example, noise from equipment operation and materials placement will impact HCP species. In addition, if pilings are used in construction, impacts from pile driving should be considered. See Section 7.1.1.1 (*Elevated Underwater Noise*) under Section 7.1 (*Dams*) for a detailed discussion of these impacts on HCP species. Second, pH alterations from the use of concrete may impact HCP species (see Section 7.1.6.5 [*Altered pH Levels*]). Finally, treated wood has been commonly used in older tide gate construction. However, this discussion assumes that new HPA projects for tide gate installation will not include treated wood, and that treated wood that is removed will be disposed of properly. If treated wood is used for new tide gate projects, then impacts on HCP species from this source should be evaluated thoroughly. For a detailed discussion of the effects of treated wood on fish and invertebrates, see the Marinas white paper (Herrera 2007c).

Equipment operation and materials placement can result in increased suspended sediments (from construction and dredging), reduced riparian vegetation, and reduced aquatic vegetation. For details on impacts on fish and invertebrates and their habitats from construction related activities, see Section 7.1.1 (*Construction and Maintenance Activities*) in Section 7.1 (*Dams*).

7.6.2 Hydraulic and Geomorphic Modifications

The presence of tide gates will impact hydraulic and geomorphic processes in a number of ways. Tide gates will alter tidal exchange by preventing free movement of saline and fresh waters in estuarine settings. Channel geometry will be changed through restriction of freshwater flow through the tide gate. Finally, substrate composition will be altered through changes in flow regime, similar to those imposed by dams. These hydraulic and geomorphic modifications are summarized in more detail below.

7.6.2.1 Altered Flow Regime–Altered Tidal/Flood Water Exchange

A tide gate is essentially an extension of a dike or a levee that allows the regulated movement of water, sediments, and organic material between river–floodplain and marine–estuarine wetland environments. Hydraulic impacts extend both upstream and downstream from a tide gate, potentially affecting a range of habitats. Overall, flow rates and flow paths are altered in the presence of a tide gate (Vandenavyle and Maynard 1994). Tide gates alter natural tidal flushing by restricting tidal flows for an unnaturally long time. In some cases, tide gates can be closed for more than 50 percent of the day (Giannico and Souder 2005). The direct and indirect impacts on

HCP species from an altered flow regime are described for dams in Section 7.1.2 (*Hydraulic and Geomorphic Modifications*) for effects on riverine habitats. The direct and indirect impacts occurring in tidal and nearshore environments, such as altered wave energy, current velocities, and nearshore circulation on HCP species, are described in Section 7.4.2.1 (*Submerged Outfalls*) and for dams in Section 7.1.2 (*Hydraulic and Geomorphic Modifications*).

7.6.2.2 Altered Channel Geometry

Tide gates may alter channel geometry in several ways. When tide gates are open, high velocities through the tide box may increase scour downstream, creating a scour pool; at the same time, water flow through the gate can entrain sediment and create a scour pool above the gate (Giannico and Souder 2005). These increased velocities are a function of the upstream–downstream differences in hydraulic head. This scour can alter the depth and width of the channel and marsh and potentially lead to habitat loss and fragmentation, as well as a loss of desirable depths if scour pools become large. When the tide gate is closed, upstream water begins to pool, increasing the channel width and depth. Although this periodic draining and inundation is a natural feature within these environments, the duration of inundation and water depths do not resemble natural conditions; therefore, the hydraulic and geomorphic modifications that occur will likely adversely affect HCP species. For a description of the impacts of altered channel geometry, see Section 7.1.2.1.2 (*Altered Channel Geometry*) in Section 7.1 (*Dams*).

7.6.2.3 Altered Substrate Composition

When a tide gate is closed, sedimentation increases landward of the structure due to slower velocities. When the gate is opened, scour downstream can occur from increased velocities (Giannico and Souder 2005; Zhang et al. 2000). The lack of two-way tidal flushing in the presence of tide gates also increases sedimentation (Anisfeld et al. 1999). Similar to a dam, a tide gate will slow water velocity upstream of the gate, as well as allow for easier settling of sediment when tidal flushing is reduced. This sedimentation can gradually convert aquatic habitats to terrestrial habitats as distributary channels and other features fill with sediment. A detailed description of the impacts in riverine environments is provided in Section 7.1.2.1.4 (*Altered Substrate Composition*) under Section 7.1 (*Dams*). For impacts in marine environments, see Section 7.4.2.1.5 (*Altered Substared Composition*) and Section 7.4.2.1 (*Submerged Outfalls*).

7.6.3 Ecosystem Fragmentation

Tide gates may cause ecosystem fragmentation from a number of different pathways. Habitat loss and fragmentation can occur from a loss of longitudinal connectivity. This alteration, combined with altered flow velocities when the gate is open, can block the migration of fishes and invertebrates (Giannico and Souder 2005). For example, blue crab showed increased difficulty navigating high flows through a tide gate in North Carolina (Rulifson and Wall 2006). Two factors may influence the extent that a tide gate blocks fish passage: the length of time the gate is open, and how wide it opens. Habitat loss also occurs from a reduction in channel–

floodplain connectivity as a result of altered channel geometry and flow regime. For a discussion of the impacts on HCP species from habitat fragmentation in riverine environments, see Section 7.1.3 (*Ecosystem Fragmentation)* in Section 7.1 (*Dams*), as well as the Fish Passage white paper (Herrera 2007a).

Ecosystem fragmentation may occur from a loss of lagoon habitat from the construction of tide gates (as well as dikes). Lagoons provide important rearing habitat for juvenile salmonids, including Chinook (Busby and Barnhart 1995) and coho salmon (Minakawa and Kraft 2005), as well as Pacific herring (Saiki and Martin 2001). This habitat can be lost due to changes in the tidal prism (Sherwood et al. 1990). Lagoons, sometimes referred to as pocket estuaries, have declined both in terms of size and number in Puget Sound due to human modifications to lowland coastal areas (Beamer et al. 2005). In fact, access to high-quality lagoon habitat has been shown to be the critical path in the restoration of Chinook salmon in the Skagit River system (Beamer et al. 2005). The primary impact of a loss of lagoon habitat would be to expose juvenile salmonids and forage fish to an increased risk of predation (Hood 2006; Wagner and Austin 1999). The Olympia oyster also uses lagoons; if a tide gate were placed near or adjacent to a lagoon, this species could be killed because of desiccation as a result of the loss of tidal inundation (Baker 1995).

In nearshore and tidal environments, ecosystem fragmentation will result from altered wave energy, current velocities, and nearshore circulation. The degree of these impacts on some HCP species would depend on the volume of the tide gate discharge, magnitude of tidal changes, and local mixing. The impacts on HCP species from altered nearshore circulation are discussed in detail in Section 7.4.2.1 (*Submerged Outfalls*).

Similar to the presence of a dam, tide gates can alter species composition and diversity. For example, Danish wetlands subjected to the influence of tide gates experienced declines in bird species diversity, declines in benthivore abundance, and increases in herbivore abundance (Holm and Clausen 2006). In addition, macrophyte biomass increased, but sea grass diversity decreased. The authors attributed many of these changes in plant communities to altered salinity levels. Seagrasses are extremely important to many of the HCP species; therefore, the loss of seagrass would result in potential impacts on many HCP fish and invertebrates (see Section 7.6.5 [*Aquatic Vegetation Modifications*] under Section 7.1 [*Dams*] for details). For a discussion of the impacts on HCP species from changes in salinity, see Section 7.6.6.6 (*Altered Salinity*).

Finally, tide gates can contribute to the loss of large woody debris. Little information is available on LWD dynamics as they are affected by tide gates and represents a potential data gap. However, information regarding impacts on large woody debris from dams in riverine ecosystems is presented in Section 7.1.3.1.4 (*Altered Large Woody Debris Transport and Recruitment*) under Section 7.1 (*Dams*).

7.6.4 Riparian Vegetation Modifications

Removal or disturbance of riparian vegetation during construction and maintenance activities associated with tide gates can adversely affect both riverine and marine environments due to the structures' location in estuarine and tidal marsh systems. Submechanisms of impact and related ecological stressors for riverine environments are discussed under dams in Section 7.1.4 (*Riparian Vegetation Modifications*).

Removal or disturbance of marine riparian vegetation during the construction of tide gates can expose HCP species to stressors caused by a variety of submechanisms of impact. These submechanisms are the same as in riverine environments and are discussed below.

7.6.4.1 Submechanisms of Impact

The submechanisms of impact from riparian vegetation removal in marine environments are the same as in riverine environments and include:

- Altered shading, solar input, and ambient air temperature
- Altered bank and shoreline stability
- Altered allochthonous inputs
- Altered groundwater–surface water interactions
- Altered habitat complexity
- Altered nutrient/pollutant loading.

The submechanisms and related ecological stressors specific to marine environments are discussed below.

7.6.4.1.1 Altered Shading, Solar Input, and Ambient Air Temperature

The influence of shade on nearshore water quality parameters such as temperature is not well established. In general, seasonal air temperature conditions, winds, currents, stratification, and tidal exchange play more dominant roles in determining marine water temperatures (Brennan and Culverwell 2004). However, shade may strongly influence temperatures in specific habitat types under specific circumstances, such as the upper intertidal zone, tidal pools, pocket estuaries (lagoons), and other habitat types that become temporarily isolated or exposed by tidal dynamics. These systems can experience increased variability in temperature and microclimate conditions in the absence of protective shading. Microclimatic conditions in the upper intertidal zone, for example, are demonstrably influenced by riparian vegetation. Rice (2006) compared microclimate parameters at a bulkheaded Puget Sound beach with no overhanging riparian vegetation to those at an adjacent unmodified site with extensive riparian vegetation. He documented significant differences in light intensity, air temperature, substrate temperature, and humidity levels at the modified site. Differences in peak substrate temperatures were particularly striking, averaging nearly 20^{o}F (11^{o}C) higher at the modified site.

7.6.4.1.2 Altered Bank and Shoreline Stability

Although it is unlikely that tide gate construction would require the removal of significant areas of riparian vegetation, many related HPA-permitted activities involve the temporary or permanent modification of the riparian vegetation structure. Riparian vegetation is an important component of the aquatic ecosystem that serves a variety of important functions for habitat structure, water quality, and biological productivity.

Marine riparian vegetation clearly plays a role in stabilizing marine shorelines, particularly bluffs and steep slopes (Brennan and Culverwell 2004; Desbonnet et al. 1994; Lemieux 2004; Myers 1993), but the specific mechanisms are not as well understood as they are in freshwater environments. The extent to which vegetation affects beach and slope stability varies depending on shoreline characteristics and the types of vegetation present (Lemieux 2004; Myers 1993). On steeper slopes, marine riparian vegetation helps to bind the soils and protect against destabilization, slides, and cave-ins that can imperil structures and disrupt the ecology of the nearshore by increasing sedimentation and burying vegetation (Brennan and Culverwell 2004). On shorelines with shallower slopes, marine riparian vegetation dissipates wave energy, thereby reducing erosion and promoting the accumulation of sediments.

7.6.4.1.3 Altered Allochthonous Input

Allochthonous inputs of organic material and large wood from marine riparian systems also have demonstrable effects on nearshore habitat conditions. While the importance of allochthonous inputs of litter is not as well documented as the linkages established for freshwater systems, its importance to marine ecosystems is nonetheless apparent (Brennan and Culverwell 2004; Lemieux 2004).

7.6.4.1.4 Altered Groundwater–Surface Water Interactions

Alteration or removal of riparian vegetation in marine environments is expected to alter groundwater inputs. While less well-studied, similar effects are likely to occur in marine systems as they do in riverine systems. Tide gates that require the removal of riparian vegetation may lead to localized increases in substrate temperature due to the loss of cool groundwater flow (Penttila 2001). A more detailed discussion of the impacts from groundwater–surface water interactions is provided under dams in Section 7.1.2.1.5 (*Altered Groundwater–Surface Water Interactions*).

7.6.4.1.5 Altered Habitat Complexity

By maintaining bank stability and contributing large wood to the aquatic environment, riparian vegetation forms and maintains habitat complexity. Driftwood and/or LWD helps to build and maintain beach habitat structure. Documented LWD functions for beach stability include its contribution to roughness and sediment trapping (Brennan and Culverwell 2004; Gonor et al. 1988) as well as inputs of organic matter, moisture, and nutrients that assist in the establishment and maintenance of dune and marsh plants (Williams and Thom 2001). Eilers (1975) found that piles of downed trees in the Nehalem (Oregon) salt marsh trapped enough sediment to support vegetation, whereby marsh islands that trapped sedge seeds provided an elevated substrate for less salt-tolerant vegetation. Herrera (2005) suggested that driftwood at the top of the beach may also slow littoral drift and reduce wave-induced erosion. It has been suggested that estuarine wood can affect water flow and the subsequent formation of bars and mudbanks (Gonor et al. 1988). The beneficial habitat structure functions of LWD along marine shorelines may be maximized if trees that fall perpendicular to beaches typically remain in place. In a recent study, local fallen trees tended to stay in place along Thurston County shorelines (Herrera 2005). The perpendicular alignment of LWD across the beach provides the LWD structure for the widest possible portion of the aquatic habitat, thus maximizing the potential area for sediment trapping and organic matter contributions.

Marine shorelines that have been modified by human activities tend to have less LWD and driftwood than unmodified beaches (Herrera 2005; Higgins et al. 2005). In particular, flow alterations from tide gates may redistribute LWD such that it concentrates in certain areas and is absent in others (Miller et al. 2001).

7.6.4.2 Direct and Indirect Effects

Riparian shade may strongly influence microclimate conditions in the upper intertidal zone. Loss of riparian shade is correlated with increased substrate temperatures and reduced humidity, which in turn are indicative of increased desiccation stress (Rice 2006). This is a significant finding because temperature and desiccation are significant stressors that limit the survival of many upper intertidal organisms, including HCP forage fish species (Brennan 2004; Brennan and Culverwell 2004). Penttila (2001) reported much higher egg mortality rates among surf smelt for eggs deposited on unshaded beaches compared to those sites with intact overhanging riparian vegetation. The hypothesized mechanism causing the observed higher rate of mortality was increased egg desiccation due to longer periods of direct sun exposure at sites with insufficient riparian vegetation to provide shade and other favorable microclimate conditions. This hypothesis is strongly supported by the findings of Rice (2006), which compared differences in microclimate conditions and surf smelt spawn survival on shaded versus unshaded beaches. The precise thermal limits for HCP invertebrates species are unknown, but it is clear that these limits exist. For example, Olympia oysters can withstand 86°F (30°C) for several hours (Baker 1995), but it is also likely that they experience diminished productivity at these temperatures.

The indirect effects on fish and invertebrates as a result of reduced bank and shoreline stability relate to the increased probability of slope failures. Although slope failures occur naturally and are an important process that maintains the proper substrate habitat of adjacent beaches (Finlayson 2006), the immediate and unnatural impacts of riparian vegetation removal can adversely affect HCP species. These effects result from the increased turbidity of adjacent waters and the potential burial of invertebrates. Effects due to increased turbidity are discussed under dams in Section 7.1.6.3 (*Altered Suspended Sediments and Turbidity*).

Marine riparian vegetation is a known source of organic matter, nutrients, and macroinvertebrate prey items for HCP species, and the recruitment of these materials is diminished when riparian vegetation is removed or modified (Brennan et al. 2004; Lemieux 2004; Maser and Sedell 1994; Miller et al. 2001; Sobocinski 2003; Williams et al. 2001). Sobocinski (2003) has documented the importance of insect communities and benthic infauna that are either a direct or indirect result of riparian vegetation. These lower trophic organisms serve as the basis of the food web for HCP fish species that use the upper nearshore environment (Williams and Thom 2001). Therefore, alterations of allochthonous inputs will change food web interactions, potentially altering basal food web resources. These changes may lead to a loss of food supply for HCP species in these environments.

The effects of modification of riparian vegetation and the effects on the structural habitat of fish have not been as well studied in marine systems as in freshwater environments. However, Sobocinski (2003) reports that salmonid food sources were directly related to the structural complexity provided by natural LWD-laden shorelines. Therefore, it is expected that the loss of such complexity will compromise the available salmonid food sources. It is uncertain what role structural complexity plays in the life-history cycle of HCP invertebrate species, if any.

7.6.5 Aquatic Vegetation Modifications

Removal or disturbance of aquatic vegetation during construction and maintenance activities associated with tide gates can adversely affect both riverine and marine environments due to the structures' location in estuarine and tidal marsh systems. These impact mechanisms and related ecological stressors for riverine environments are discussed under dams in Section 7.1.5 (*Aquatic Vegetation Modifications*) and include altered autochthonous inputs, altered habitat complexity, and altered nutrient/pollutant cycling.

Aquatic vegetation also plays an important role in marine systems. The basis for nearly all life in the sea is the photosynthetic activity of aquatic autotrophs such as algae, cyanobacteria, benthic microalgae, benthic macroalgae (kelps and seaweeds), and seed plants (such as seagrasses, mangroves, and saltmarsh plants) (Nybakken and Bertness 2005). The availability of light is a crucial parameter for seagrasses and other aquatic autotrophs (Hall et al. 1999), although other factors (e.g., substrate type) can also play a role in the survivability of aquatic plants (Koch 2001).

Aquatic vegetation, in particular eelgrass, provides important cover for juvenile fish and invertebrates (Phillips 1984). Eelgrass also provides a necessary structural surface for a community of epibenthic organisms, making eelgrass communities one of the most productive ecotones in the Pacific Northwest (Ferraro and Cole 2007). Observations of eelgrass usage in Southeast Alaska show that the most abundant species using eelgrass meadows were chum salmon, Pacific herring, pink salmon, coho salmon, and Pacific sand lance (Johnson and Thedinga 2005).

7.6.5.1 Direct and Indirect Effects

Marine littoral vegetation is important for the colonization of organisms that are important prey resources for HCP species, such as Newcomb's littorine snail, Pacific sand lance, Pacific herring, Pacific cod, northern abalone, surf smelt, steelhead and coastal cutthroat trout, salmon (pink, chum, coho, and Chinook), Olympia oyster, bull trout, Dolly Varden, rockfish, longfin smelt, eulachon; and walleye pollock (Chambers et al. 1999; Gardner 1981; Goetz et al. 2004; Johnson et al. 1999; Larsen et al. 1995; Myers et al. 1998; Pauley et al. 1988; WDNR 2006a, 2006b; West et al. 1994).

In studies of outmigrating juvenile chum salmon in Hood Canal, Simenstad et al. (1980) found that juvenile chum fry (1.2–1.8 in [30–45 mm]) fed extensively on small, densely distributed harpacticoid copepods, and selected the largest copepods available. Similarly, Miller et al. (1976) reported that juvenile chum fed predominantly on epibenthic harpacticoid copepods. As the fish grew in size, their diet content was composed more of larger epibenthos and pelagic crustaceans. Consistent with other studies, the highest densities of harpacticoid copepods occurred in magnitudes 4–5 times higher in eelgrass stands than in sand habitat without eelgrass.

Similarly, in a study of the Drayton Harbor (Washington) marina, Thom et al. (1989) reported that juvenile salmon density was highest at the eelgrass habitat site that also supported the highest salmon prey density and epibenthos density. Similarly, total fish density increased dramatically immediately following a peak in maximum epibenthos and the most rapid increase in Zostera biomass (Thom et al. 1989). These epibenthic prey assemblages of copepods, such as the harpacticoids, are known to feed on bacteria, epiphytes, plant detritus, and diatoms. It is consistently documented that vegetation assemblages associated with eelgrass, in particular, support increased magnitudes of juvenile salmonid epibenthic prey (Cordell 1986; Simenstad et al. 1980; Thom et al. 1989).

The limitation of habitat for key prey resources likely affects migration patterns and the survival of many juvenile fish species. For smaller fish less than 1.97 in (50 mm) in length, residence times along particular shorelines are thought to be a function of prey abundance (Simenstad et al. 1980).

Eelgrass also plays a role in protecting invertebrates from both fish and avian predators (Bostrom and Mattila 1999). It is uncertain what role eelgrass plays in the protection of HCP invertebrate species, but the generality of the existing work in the field would suggest that a loss of eelgrass would increase predation of those species. As a result, any alteration of eelgrass through tide gate operation will directly affect HCP species.

7.6.6 Water Quality Modifications

Water quality modifications from tide gates are the same as those associated with dams. In general, these modifications alter five primary water quality variables: temperature, dissolved oxygen, suspended solids (turbidity), pH levels, nutrient loading, and salinity.

7.6.6.1 Altered Temperature Regime

Each species requires a certain range of temperatures for optimal survival, and alteration of natural thermal regimes will adversely affect HCP species. Abrupt changes in temperature can form as a result of blocked tidal flushing and represent a thermal barrier for migration, similar to a dam (Giannico and Souder 2005). A detailed discussion of the impacts on fish and invertebrates is provided under dams in Section 7.1.6.1 (*Altered Temperature Regime*).

7.6.6.2 Altered Dissolved Oxygen

Disruption of natural flow can cause stratification and depletion of oxygen, with the downstream side of the tide gate becoming anoxic at the bottom (Winn and Knott 1992). In Cape Cod (Massachusetts), periodic low oxygen levels can result in large fish kills (Portnoy 1991) in tidal marsh systems.

Altered flow regime can affect dissolved oxygen concentrations through changes in soil chemistry. Normally, soils are kept under anaerobic conditions because they are inundated by tidal waters. When tidal water is excluded, soils are exposed to the air and can become aerobic. Subsequently, the exclusion of salt water can lead to oxygen depletion in the water when organic matter in the soils begins to oxidize (Giannico and Souder 2005). A detailed discussion of the impacts of depleted oxygen is provided under dams in Section 7.1.6.2 (*Altered Dissolved Oxygen*). Oxidation of peat soils can cause the level of a marsh to fall and to become compacted (Roman et al. 1984). In addition, lowered dissolved oxygen concentrations will alter redox conditions of the soils, altering pH levels and increasing metal leaching from soils. This phenomenon has been well documented in the literature. Episodic acidification of estuarine waters from the drainage of sulfate floodplain sediments is common (Anisfeld and Benoit 1997; Johnston et al. 2005a; Sammut et al. 1996). Drainage promotes the oxidation and export of sulfuric acid, a lowering of pH levels, and can result in the release of iron, lead, aluminum, copper, silver, and cadmium (Giannico and Souder 2005). In some cases, lowering pH produces iron flocs that can precipitate out of solution and cover the benthos (Sammut et al. 1996) and kill marsh plants (Giannico and Souder 2005). A detailed discussion of the impacts from metals toxicity is provided in Section 7.4.6.1 (*Metal Toxicity*) in Section 7.4 (*Outfalls*).

7.6.6.3 Altered Suspended Solids and Turbidity

Tide gates alter natural flow regimes and change natural sedimentation patterns. In addition, high velocities through open flood gates will increase erosion both up- and downstream, increasing turbidity in the downstream water. Impacts on fish and invertebrates from increased suspended sediments are described under dams in Section 7.1.6.3 (*Altered Suspended Sediments and Turbidity*).

7.6.6.4 Altered pH levels

Besides alterations from concrete use during construction, pH can be altered as a result of changes in dissolved oxygen and redox conditions. This mechanism is described in Section 7.6.6.2 (*Altered Dissolved Oxygen*). Impacts on fish and invertebrates from altered pH levels are discussed under dams in Section 7.1.6.5 (*Altered pH Levels*).

7.6.6.5 Altered Nutrient Loading

Changes in nutrients stem from removal of vegetation and delivery of nutrients to tidal marshes from upland areas. Impacts from nutrient loading are discussed under dams in Section 7.1.6.6 (*Altered Nutrient Loading*).

7.6.6.6 Altered Salinity

One type of water quality alteration common with tide gates but not yet discussed in this white paper is a change in salinity. As tide gates block the movement of salt water, they will inherently change salinity both upstream and downstream of the gate. In a natural estuarine system, salinity fluctuates daily and seasonally from tides (Giannico and Souder 2005), and the presence of a tide gate will alter the natural flushing pattern. This alteration will cause a displaced salt wedge to migrate upriver (Vandenavyle and Maynard 1994). Salt water is more dense, and when a tide gate is closed, salt water settles and will migrate upstream. In addition, because water that builds up behind a tide gate is usually fresh, this pulse of fresh water is released downstream, lowering salinity in the receiving water (Williams and Thom 2001). Altered salinities can cause marsh community shifts; when tide gates are present, salinity gradients are sharp and can delay the migration of fish (Pearlstine et al. 1993). Salinity is also altered in the groundwater environment. In Australia, saltwater seepage into the surrounding groundwater was observed. Depending on the soil properties, this seepage was less than 33 to more than 262 ft (10 to more than 80 m) from the impounded area (Johnston et al. 2005b). This saltwater intrusion could have devastating effects on riparian vegetation, leading to increased bank failures, increased temperatures, and reduced nutrient cycling. The direct and indirect effects from loss of riparian vegetation are discussed in Section 7.6.4 (*Riparian Vegetation Modifications*).

7.6.6.6.1 Direct and Indirect Effects

The direct effects on fish from altered salinities are related to delayed migration. For example, juvenile salmon need a gradual change in salinity as they undergo the physiological changes needed to migrate into salt water (Groot and Margolis 1991). In can take 2–3 days (or much longer) for Atlantic salmon to reorient themselves after sudden salinity changes (Russel et al. 1998). When faced with abrupt changes in salinity, migrating fish slow down and predation could increase.

Altered salinity can also influence spawning and egg development in fish species. Stripped bass have shown a preference for low salinity (0.5 ppt or less) for spawning. In the Savannah River estuary (Georgia), striped bass have shown recruitment failure because eggs were in areas of higher salinity from tide gate operations (Vandenavyle and Maynard 1994). In laboratory experiments, striped bass eggs died within 24 hours at salinities greater than 18 ppt, and larvae exposed to salinities of 15 ppt and higher exhibited stunted growth and lower survival (Winger and Lasier 1994). In a study of Puget Sound lingcod, Cook et al. (2005) showed that the optimal salinity range was 20–30 ppt for incubation of eggs, and deformities were observed at both 15 and 35 ppt. For Pacific herring, the optimum range for development and fertilization was in the range of 4–8 ppt salinity (Griffin et al. 1998). Finally, Snake River cutthroat trout show significant mortality at 18 ppt, while a southern Bonneville stock showed higher tolerance and no mortality until 22 ppt (Wagner et al. 2001).

Overall, changes in salinity can result in delayed migration, increased predation, and mortality of developing eggs and larvae. The effects of altered salinity from tide gates on invertebrates are largely unknown.

8.0 Cumulative Effects

Evidence increasingly indicates that the most devastating environmental effects are likely not the direct effects of a particular action, but the combination of individually minor effects of multiple actions over time (CEQ 1997). Each of the flow control structures presented in this white paper will have cumulative effect ramifications. In general, as the number of flow control structures increases in a given area, impacts will accrue that increase habitat loss, alter the flow regime, and shift the composition and diversity of species. For example, tide gates are often constructed in areas converted for agriculture. As a result, irrigation that routes diversions and runoff from fields through outfalls are likely. This section assesses the cumulative effects that each of the subactivity types may have on the HCP species.

8.1 Dams

Cumulative effects from dams are well known. The presence of a dam alters stream temperatures, dissolved oxygen concentrations, nutrient loading, natural sediment transport, channel geometry, flow regime, habitat connectivity, and changes in species composition that result in cumulative impacts on HCP species. If only one of these impacts were realized, the impacts may be minor; however, taken in concert, these impacts can overwhelm some species and negatively affect their survival, growth, or fitness.

A series of dams on a given river or river system will compound difficulties for migrating species. For example, in a study on the Columbia River, only 3 percent of tagged Pacific lamprey reached the most upstream site of a series of 3 dams (Moser et al. 2002). However, 40–50 percent of them passed over the lower dams, indicating that as the number of structures increase, successful migration to the upper reaches of a watershed will decrease. In addition, declines in Columbia River salmon and steelhead were the result of cumulative impacts from nine hydropower dams on the mainstem, each contributing 2–20 percent of the overall loss (Williams and Thom 2001). From a geomorphic standpoint, a series of dams will compound sediment losses to downstream coastal systems, exacerbating beach loss and erosion. In terms of eutrophication, nutrient loading from several dams may lead to the development of low-oxygen zones in coastal areas.

In many cases, these cumulative impacts extend well beyond the location of the dam. For example, in the highly impounded Columbia River watershed, effects from dams high in the watershed will translate to the marine environment. On the Olympic Peninsula, the Elwha River dams are causing significant beach losses from sediment accumulation in reservoirs behind two large dams (DOI 1995).

8.2 Weirs

The cumulative effects from weirs on HCP species are similar to those described above for dams. However, these impacts are lessened due to the scale of weir projects and the fact that these are overflow structures with fewer impacts on the downstream water quality.

8.3 Dikes and Levees

Dikes and levees alter channel geometry, flow regime, and habitat connectivity, contributing to cumulative effects on HCP species. As with most flow control structures, the more levees constructed in a given area, the more fragmentation of the habitat will result. In addition, the presence of several dikes and levees in a watershed will compound the effects of flow changes downstream. For example, a given increase in flood flow from one channelized reach flowing into another such reach will increase the peak flood flows because there will be an increased amount of disconnected floodplain area. Normally, the floodplain would be able to absorb these flood flows and to minimize the downstream effects of peak flows.

8.4 Outfalls

Limited information is available regarding the cumulative impacts of hydraulic and geomorphic modifications associated with outfall structures. However, a string of poorly designed outfalls could easily starve a shoreline of sediment, just as groins have done in other parts of the world (Byrnes and Hiland 1995). If riparian vegetation is removed during the construction of an outfall, changes in temperature and solar input will be magnified as more such outfalls are placed within a watershed. Similarly, water quality degradation from a single outfall might be minimal; however, the more outfalls that are located in a single stream reach, the more likely it is that impacts will occur on HCP species from metals toxicity, low oxygen, and exposure to organic pollutants.

8.5 Intakes and Diversions

As with outfalls, limited information is available regarding the cumulative impacts of hydraulic and geomorphic modifications associated with intakes and diversion infrastructure. Intakes have specific modifications that could have significant cumulative impacts. In particular, their design does not adequately account for the entrainment of spawn and drifting larvae along river system. This type of cumulative impact has been described in terms of large-scale hydropower planning in Europe (Larinier 1998). If riparian vegetation is removed during construction of an intake, changes in temperature and solar input will be magnified as more outfalls are placed within a watershed. In addition, as more diversions are located within a watershed, the more of an impact will occur on the downstream flow regime. An extreme situation could result in a completely dry channel from multiple diversions, which would make the river reach unusable for HCP species.

8.6 Tide Gates

The cumulative effects from tide gates are similar to those for a dam. Because tide gates block migration and tidal flows, the more tide gates are present in a given area, the more impacts on HCP species would occur. These cumulative impacts translate to water quality modifications as well. For example, changes in salinity are a fundamental impact from the presence of a tide gate. The more tide gates there are in a system, the greater this impact will become. Changes in salinity are important to migration patterns and to provide suitable habitat for species that use these areas. In addition, metals toxicity from altered flow, oxidation of marsh soils, and changes in pH will be compounded if several tide gates are located within a given area.

Cumulative effects from saltwater intrusion into the riparian zone may also develop. In Australia, it was observed that saltwater seepage into the surrounding groundwater occurred. Depending on soil properties, this seepage was less than 33 ft to more than 262 ft (10 m to more than 80 m) from the impounded area (Johnston et al. 2005b). This saltwater intrusion could have a devastating effect on riparian vegetation, leading to increased bank failures, increased temperatures, and reduced nutrient cycling.

9.0 Potential Risk of Take

Flow control projects are typically designed with the intent of withdrawing water and/or modifying the hydraulic and hydrologic characteristics to promote human uses of the aquatic environment and the surrounding landscape. Given their intended purpose, these projects lead to a fundamental alteration of ecological processes. Therefore, they impose a range of direct and indirect effects on the environment, resulting in an array of ecological stressors, during both the construction phase and over the course of operation. The magnitude of these stressors will vary depending on the scale of the project in question and the degree to which it modifies ecological conditions and processes.

This section provides a narrative and tabular summary of the species risk of take resulting from exposure to impact mechanisms and related ecological stressors associated with the construction and operation of flow control structures. For the purpose of this analysis, flow control structures include the following subactivity types: Dams; weirs; dikes and levees; outfalls; diversion structures and water intakes; and tide gates. This summary is derived from the impact mechanism and stressor specific risk of take ratings developed for the 52 HCP species in the exposure–response matrices, which are presented in Appendix A. The risk of take assessment, presented in Appendix A and summarized here, was developed based on the likelihood of exposure for each of the 52 HCP species to the impact mechanisms and stressors imposed by each subactivity type, as well as the sensitivity of exposed life-history stages to these stressors.

The risk of take is rated by impact mechanism for each species (or species grouping, as listed in Table 1-1) using the criteria presented in Table 6-3. As noted, the risk of take rating criteria are based on the following assumptions:

- **High risk of take (H)** ratings are associated with:
    - ☐ Stressor exposure is likely to occur with high likelihood of individual take in the form of direct mortality, injury, and/or direct or indirect effects on long-term survival, growth, and fitness potential due to long-term or permanent alteration of habitat capacity or characteristics. Likely to equate to a Likely to Adversely Affect (LTAA) finding.
- **Moderate risk of take (M)** ratings are associated with:
    - ☐ Stressor exposure is likely to occur, causing take in the form of direct or indirect effects potentially leading to reductions in individual survival, growth, and fitness due to short-term to intermediate-term alteration of habitat characteristics. May equate to an LTAA or a Not Likely to Adversely Affect (NLTAA) finding depending on specific circumstances.

- **Low risk of take (L)** ratings are associated with:
  - ☐ Stressor exposure is likely to occur, causing take in the form of temporary disturbance and minor behavioral alteration. Likely to equate to an NLTAA finding.
- **Insignificant or discountable risk of take (I)** ratings apply to:
  - ☐ Stressor exposure may potentially occur, but the likelihood is discountable and/or the effects of stressor exposure are insignificant. Likely to equate to an NLTAA finding.
- **No risk of take (N)** ratings apply to species with no likelihood of stressor exposure because they do not occur in habitats that are suitable for the subactivity type in question, or the impact mechanisms caused by the subactivity type will not produce environmental stressors.
- **Unknown risk of take (?)** ratings apply to cases where insufficient data are available to determine the probability of exposure or to assess stressor response.

The species risk of take summary is organized by subactivity type, impact mechanism category, and environment type. In the following subsections, a general description of the risk of take associated with each subactivity type is provided by impact mechanism category and, where appropriate, by submechanism. In cases where the physical effects and related risk of take are similar between environment types, the risk of take discussion is grouped to avoid redundancy. The narrative summary for each subactivity type is supported by a risk of take matrix, identifying the overall risk of take for each of the 52 HCP species. Risk of take is rated for each species by impact mechanism category and environment type (i.e., riverine, marine, and lacustrine) (Tables 9-1 through 9-6, presented at the end of the narrative). The summary risk of take presented in the narrative and the matrices for each impact mechanism category represents the greatest overall risk of take from each of the submechanism of impact in that category.

9.1 Dams

Dams are a significant form of hydromodification that impose broad and pervasive effects on riverine environments. Dam projects range in scale from the relatively modest on small stream systems to immense projects on large river systems, such as the Mossy Rock Dam on the Cowlitz River or the Grand Coulee Dam on the Columbia River. Dams are, by definition, channel spanning structures that create upstream impoundments. These structures impose stressors on aquatic organisms through a range of impact mechanisms and fundamentally alter the characteristics of riverine ecosystems, and in some cases lacustrine ecosystems (e.g., where dams are created at lake outlets). The hydrologic and water quality effects of dams can extend to marine ecosystems as well. For the purpose of this white paper, the species risk of take associated with dams focuses on the effects of this subactivity type on riverine and lacustrine environments.

Species-specific risk of take ratings for dam development, operation, and removal are presented by impact mechanism in Table 9-1. The specific stressors and related species risk of take from impact mechanisms caused by this subactivity type are described in the following subsections.

9.1.1 Construction and Maintenance

Construction, operation, and maintenance of dams involve a diverse array of activities that can impose a variety of environmental stressors on HCP species occurring in riverine and lacustrine environments. Construction and maintenance may include such activities as heavy equipment operation, materials placement, pile driving, and flow bypass and dewatering around work areas. The majority of construction and maintenance activities are temporary in nature, lasting from a few days to several weeks, depending on the size of the project and the nature of the activity. In the case of large dams, however, construction and maintenance activities may last for months or even years, with continuous activity occurring throughout. The risk of take associated with construction activity varies by impact mechanism and is dependent on the project-specific magnitude of that impact mechanism. As discussed below, some mechanisms may produce a high risk of individual take due to their intensity, while others may result in a low risk of take due to their limited magnitude and duration.

Construction-related effects during dam removal must also be considered. Many of the activities associated with dam removal, such as equipment use, materials placement, and visual, noise, and physical disturbance, are similar to those imposed during construction. However, the dewatering of impoundments creates the potential for unique effects in the form of stranding in dewatered areas that must be considered when evaluating risk of take.

The species-specific risk of take ratings for dam-related construction and maintenance impacts reflects this perspective. These ratings are presented at the end of this narrative in Table 9-1. The risk of take resulting from each associated submechanism of impact is described in further detail below.

9.1.1.1 Elevated Underwater Noise

The construction, operation, and maintenance of dams will result in some alteration of the underwater noise environment. The nature of this habitat modification will vary depending on the phase of the project. During construction and maintenance, intense sources of underwater noise such as pile driving, materials placement, or in-water equipment operation may create short-term pulses of high intensity sound pressure. Sound pressure waves of sufficient intensity from sources such as pile driving have been shown to cause injury or even direct mortality of fish and invertebrates in laboratory tests and *in situ*. Underwater noise insufficient to cause injury or mortality may still lead to behavioral modifications (e.g., startle or avoidance responses), which increase stress, alter feeding patterns, and may decrease the ability to avoid predators. Auditory masking effects caused by continuous noise sources that alter the ambient noise level (e.g., from extended operation of construction and maintenance vessels, in-water equipment use, or spillway and turbine operation) may affect the ability of fish to detect predators and prey, affecting their survival, growth, and productivity.

Specific information on the risk of take associated with underwater noise is relatively limited. Of the possible sources of underwater noise, pile driving, which may be a component of dam construction, has been subjected to the most research. A sufficient base of information on the subject has been assembled to establish effects thresholds for disturbance and injury in HCP salmonid species. These thresholds provide a useful basis of comparison for qualitatively estimating the effects of other sources of underwater noise.

Until recently, NOAA Fisheries and USFWS recognized underwater noise levels of 150 dB_{RMS} and 180 dB_{peak} as thresholds for disturbance and injury, respectively, of federally listed salmonid species (Stadler 2007; Teachout 2007). While the disturbance threshold still stands, on April 30, 2007, NOAA Fisheries established the following dual criteria to evaluate the onset of physical injury to fishes exposed to underwater noise from impact hammer pile driving (NMFS 2007b):

> ***SEL:*** *A fish receiving an accumulated Sound Exposure Level (SEL) at or above 187 dB re: one micropascal squared-second during the driving of piles likely results in the onset of physical injury; a simple accumulation method shall be used to sum the energy produced during multiple hammer strikes.*
>
> ***Peak SPL:*** *A fish receiving a peak sound pressure level (SPL) at or above 208 dB re: one micropascal (which equates to* dB_{peak}*) from a single hammer strike likely results in the onset of physical injury.*

Exceeding either criterion equals injury. Pile-driving related noise has received the most scrutiny because it produces stressors of the greatest magnitude. The effects thresholds established for pile driving provide a useful basis for a qualitative estimation of the effects of other sources of underwater noise. While data on other sources of underwater noise are limited, the data available indicate that these noise sources are unlikely to exceed established injury thresholds. However, many sources of underwater noise associated with dam construction, maintenance, and operation may well exceed disturbance thresholds and/or produce noise of sufficient magnitude to cause auditory masking effects. Therefore, projects involving some element of in-water work are expected, at the least, to result in a moderate risk of take from altered underwater noise conditions. When considering this potential, however, the change in ambient noise levels must be measured against the existing riverine background noise levels. Unfortunately, little reference data are available on background noise levels in riverine environments in the existing literature. Regardless of existing noise levels, projects involving pile driving during construction and maintenance would be equated with a high risk of take.

9.1.1.2 Equipment Operation and Materials Placement

The construction of dams requires the operation of heavy equipment and the placement of materials in and around aquatic habitats and adjacent terrestrial habitats, including riparian zones and floodplains. In-water use of equipment and the placement of materials impose stressors in the form of physical and visual disturbance. The magnitude of these stressors will vary widely, depending on the scale of the project in question and the specific construction measures used. Applying a worst-case-scenario perspective, the magnitude of these stressors can be significant.

Visual and physical disturbance during construction would be expected to alter fish behavior, causing temporary avoidance and startle responses, compelling individuals to move out of affected habitats or to assume a cryptic posture. Such disturbances will increase stress and exertion, may alter spawning and foraging behavior, or increase the risk of predation if fish are startled away from protective habitat. These effects may lead to decreased survival, growth, fitness, and spawning success, which equates to a moderate risk of take. Sessile, cryptic, or otherwise non-motile species or life-history stages are a possible exception. Non-motile life-history stages of individuals (e.g., salmon eggs, buried lamprey ammocoetes, mussels or snails, or juvenile salmonids hiding in interstices) face increased risk of mechanical injury or death if they are crushed or buried during construction and maintenance, which constitutes a high risk of take.

9.1.1.3 Bank/Channel/Shoreline Disturbance

Construction-related bank, channel, and shoreline disturbance could result in decreased stream bank and shoreline stability, as well as increased erosion and turbidity. These effects are localized and would occur during construction and possibly again during seasonal high-flow conditions. The risk of take associated with this stressor varies depending on species-specific sensitivity to increased turbidity. In general, more motile fish species would be expected to experience only temporary behavioral alteration and low risk of take. In contrast, less motile fish life-history stages or sessile invertebrates could experience a high risk of take due to mortality caused by smothering, as well as decreased growth and fitness due to the effects of high turbidity on foraging success.

9.1.1.4 Dewatering, Flow Bypass, and Fish Handling

Temporary dewatering and flow bypass with fish removal and relocation from work areas is a common and necessary practice during dam construction and maintenance. Even when dewatering is not required for construction and maintenance, exclusion areas are often created around the work sites to contain sediments and other pollutants and to reduce the magnitude of stressor exposure. This construction and maintenance activity poses a relatively high risk of take. Well-designed protocols and trained personnel are necessary to avoid high levels of mortality. Even with appropriate protocols and experienced field crews, high levels of mortality can result. For example, NOAA Fisheries evaluated take associated with dewatering and fish handling in a recent biological opinion. They estimated that salmonid mortality rates in the range of 8 to as high as 20 percent may occur even when trained personnel are used, and have assumed an injury rate of 25 percent (NMFS 2006).

Mortality rates may be even higher in areas with complex substrate and bathymetry. During the egg, larval, or juvenile life-history stage of many species, individuals may be too small or too cryptic to collect and relocate effectively (e.g., juvenile salmonids hiding in cobble interstices, river lamprey ammocoetes buried in fine substrate, larval or juvenile dace). Mortality is the expected outcome for any individuals stranded within the exclusion area. Even in the absence of mortality, fish handling and relocation may result in stress and injury, as well as increased competition for forage and refuge in the relocation habitat. Moreover, the act of capture, handling, or forced behavioral modification of an ESA-listed species constitutes harassment, which is considered a form of take. Thus, the permitting of channel and work area dewatering poses a high risk of take of varying levels of severity depending on habitat and species and life-history stage-specific factors.

In addition to these effects, the act of dewatering the stream and redirecting flow may pose a barrier to fish migration. Delays in migration can lead to adverse effects on spawning fitness, can increase exposure to predation and poaching, and can deny juvenile fish access to rearing habitats during critical periods. These effects constitute a moderate risk of take of HCP species with migratory life-history stages.

9.1.1.5 Dewatering Associated with Dam Removal

Once a dam is breached, the impoundment behind the structure will drain. Aquatic species in the impoundment trapped in rapidly dewatering habitats face risk of mortality from stranding, particularly non-motile species and life-history stages. Motile species able to avoid stranding will be displaced from existing habitats and forced to relocate within disturbed habitats that may present limited foraging opportunities, which could similarly limit survival, growth, and fitness. It is generally presumed that care will be taken during dam removal to dewater slowly, reducing stranding risk. Consistent with a worst-case scenario approach, however, this activity must be associated with a high risk of take, particularly for non-motile species and life-history stages that may be exposed to this stressor.

9.1.2 Hydraulic and Geomorphic Modifications

Dams impose significant changes in the hydraulic and geomorphic characteristics of riverine and lacustrine environments. These modifications can, in turn, significantly modify the characteristics and suitability of the affected habitats for HCP species adapted to riverine environments. The submechanisms through which these impacts manifest are complex, even before considering the complexity of the responses of HCP species to stressor exposure. Therefore, for the purpose of this analysis, the risk of take associated with these submechanisms is viewed in a holistic fashion.

The species-specific risk of take ratings resulting from dam-related hydraulic and geomorphic modification (as presented in Table 9-1) reflect this holistic perspective. The basis for these ratings is discussed by submechanism in the following sections.

9.1.2.1 Altered Flow Regime, Channel Geometry, and Substrate Composition

Flow regime, channel geometry, and substrate composition and stability are dominant factors determining aquatic habitat structure in riverine environments. Alteration of any of these habitat components can change the suitability of the habitat for various life-history stages of HCP species. These habitat alterations are essentially permanent and continuous and can lead to changes in the productivity of the habitat for spawning, foraging, rearing, and refuge. In a worst-case scenario, these effects in turn are likely to lead to reduced spawning success, as well as reduced survival, growth, and fitness for species and life-history stages dependent on the affected habitat.

Dams fundamentally alter these environmental characteristics by converting a flowing water environment upstream of the structure to a slack water impoundment, altering the hydrologic regime and interrupting the transport of wood, sediment, and organic material. Upstream of the structure, historically lotic habitat is converted to a lentic impoundment. Downstream of the structure, alteration of flow regime and reduced transport of LWD and sediment from upstream sources are likely to lead to changes in channel morphology, with detrimental effects on habitat structure. Operational water level fluctuations may also affect habitat productivity, creating risk of stranding for non-motile fish life-history stages and invertebrates, which is likely to lead to mortality. The effects that dams impose on ecosystem structure and function through these submechanisms are interrelated, as is the risk of take. These collective effects have been shown to alter habitat suitability for fish and invertebrate species adapted to the original environmental condition. A broad array of research has demonstrated the effects of these individual submechanisms on the survival, growth, and fitness of many of the HCP species that occur in riverine environments. In some cases, these effects have been shown to limit productivity at the population level, depending on the nature of the facility and the species affected. Using the criteria defined for the purpose of this white paper, the effects on survival, growth, fitness, and productivity caused by long-term alteration of environmental and habitat characteristics equates to a high risk of take.

9.1.2.2 Altered Groundwater–Surface Water Interactions

The effects that dams impose on the connectivity between surface water and groundwater are complex and change over time. Most dams are designed to be relatively impermeable at their base to prevent the loss of impounded water to groundwater. However the large hydraulic head created by dams can, in some cases, increase groundwater exchange, resulting in increased hyporheic flow to downstream reaches. Over time, however, the accumulation of fine sediments in the impoundment will decrease bed permeability and retard groundwater exchange. Besides altering connection between surface and groundwaters, changes in flow regime, sediment transport, and substrate composition will all affect in-channel hyporheic exchange as well.

Hyporheic exchange is an important component of ecosystem function (including water quality moderation) in riverine environments. Therefore, this impact mechanism has the potential to affect juvenile and/or adult survival, growth, and fitness, and in some cases the spawning productivity of a range of species. Using the criteria defined for the purpose of this white paper, the effects on survival, growth, fitness, and productivity caused by long-term alteration of environmental and habitat characteristics equate to a high risk of take.

Species with a moderate risk of take include those with life-history stages that are dependent on hyporheic exchange for its beneficial effects on water temperature and dissolved oxygen levels. For example, most salmonids preferentially spawn in areas with groundwater-induced upwelling, which promotes oxygenation of spawning gravels. Alteration of hyporheic exchange in environments suitable for spawning could potentially affect egg survival and reduce the availability of suitable spawning habitat, resulting in reduced spawning success. Similarly, groundwater inflow can provide important thermal refugia for migrating adult and rearing juvenile salmonids during periods with high water temperatures. A reduction in the amount of thermal refugia may negatively affect survival during these life-history stages. Similar effects would be expected for other coldwater fish species with low thermal tolerance thresholds, such as pygmy whitefish. More generally, hyporheic exchange also plays a key role in nutrient cycling and food web productivity in alluvial bed rivers. Projects resulting in significant alteration of hyporheic exchange could adversely affect food web productivity, limiting foraging opportunities for fish and invertebrate species dependent on these types of environments.

9.1.3 Ecosystem Fragmentation

Ecosystem fragmentation is an impact mechanism that incorporates the collective effects of the loss of habitat within the footprint of the structure, and the resulting effects of hydraulic and geomorphic modification imposed by the structure on the environment. These include effects on the migration and dispersal of organisms; the transport, distribution, and biogeochemical processing of LWD and other organic material, and changes in the introduction, cycling and sequestration of nutrients and pollutants. As with hydraulic and geomorphic modification, this impact mechanism is a significant and multifaceted component of the effects that dams impose on the aquatic environment.

The species-specific risk of take associated with ecosystem fragmentation caused by dams is presented by environment type in Table 9-1.

9.1.3.1 Altered Longitudinal Connectivity

The predominant effect of dams is the fragmentation of longitudinal connectivity of the river continuum. Dams interrupt the downstream transport of water, wood, sediment, and organic material, and, depending on design and scale, may also prevent the upstream and downstream movements of migratory fish and invertebrates. The impoundment also creates a lentic habitat that is discontinuous within the riverine landscape, capable of altering water quality (e.g., temperature and nutrient loading) and food web productivity.

Using the criteria defined for the purpose of this white paper, the effects on survival, growth, fitness, and productivity caused by long-term alterations of environmental and habitat characteristics equate to a high risk of take.

9.1.3.2 Altered River–Floodplain Connectivity

Dams can cause a significant alteration in the connectivity of the river system to floodplain and terrestrial habitats. In the impoundment, the changes are obvious. The channel, floodplain, and portions of the surrounding valley are inundated. Depending on site-specific topography, the natural gradient between the river and floodplain is replaced by a steeper ecological gradient between the new aquatic and surrounding terrestrial habitat. This gradient may be quite abrupt if impoundment management causes extreme water level fluctuations, creating simplified habitat conditions at the impoundment margin that are not suitable for rearing, spawning, refuge, or other important life-history requirements.

In downstream habitats, changes in flow regime and sediment starvation may lead to channel degradation, causing fragmentation of the main channel from off-channel and floodplain habitats. The connectivity between river and floodplain habitats is reduced over a broad range of flow conditions. As discussed in the Habitat Modifications white paper (Herrera 2007e), the implications of this degraded connectivity are significant for ecosystem productivity. A number of HCP species are dependent on off-channel and floodplain habitats during one or more life-history stage. A reduction in the availability of suitable habitat will lead to increased competition for the remaining available habitat, decreased growth and fitness, increased exposure to predation, and potentially decreased availability of suitable spawning sites. While these effects primarily concern fish, invertebrate species such as mussels would also be affected due to reduced productivity of host fish populations.

Using the criteria defined for the purpose of this white paper, the effects on survival, growth, fitness, and productivity caused by long-term alteration of environmental and habitat characteristics equate to a high risk of take.

9.1.3.3 Altered LWD Transport and Recruitment

The influence of LWD on riverine habitat complexity is broadly recognized. Dams interrupt the transport of LWD along the longitudinal gradient in riverine environments. Modification of the flow regime in downstream reaches and channel downcutting caused by sediment starvation may also lead to lateral river-floodplain fragmentation, which could limit the recruitment in downstream reaches, further starving the channel of LWD. The hydraulic and geomorphic effects of reduced LWD density in the channel network can lead to further alterations in habitat complexity. This impact mechanism presents a potential risk of take for a broad range of species dependent on riverine aquatic ecosystems through a variety of species-specific stressors. Depending on the particular life history of the affected species, alterations in habitat complexity may limit the availability of suitable spawning, resting, and rearing habitat, and may alter foraging opportunities and predation exposure. In general, fish species that are dependent on habitats potentially affected through this mechanism of impact are likely to experience decreased spawning success and/or decreased survival, growth, and fitness due to an overall reduction in suitable habitat area. Using the criteria defined for the purpose of this white paper, and because these effects are imposed by long-term alteration of hydraulic and geomorphic characteristics, they equate to a high risk of take.

9.1.3.4 Altered Groundwater–Surface Water Interactions

The hyporheic zone is the zone of vertical ecological connectivity between surface water and groundwater. The effects of dams on this component of ecological connectivity and related risk of take are discussed above in Section 9.1.2.2 (*Altered Groundwater-Surface Water Interactions*) under *Hydraulic and Geomorphic Modifications* (Section 9.1.2).

9.1.3.5 Altered Community Composition

The conversion of riverine habitats from lotic to lentic environments upstream of dams, and alterations of flow and thermal regime both upstream and downstream of the structure can lead to changes in community composition within the riverine ecosystem. By creating lentic habitats and altering downstream habitat complexity and water quality conditions, dams may create suitable conditions for a range of species that would not otherwise be able to survive in the undisturbed system. For example, impoundments create warm water habitats that promote the growth of emergent vegetation, creating habitat conditions suitable for warm water fish (e.g., bass, perch, and sunfish) that would not normally survive in a flowing river with naturally cool temperatures. These species may compete with juvenile salmonids for food resources, or may prey on them directly, affecting their survival, growth, and productivity. By causing reductions in downstream habitat complexity and interrupting the transport of coarse particulate organic matter, dams may indirectly cause a shift in macroinvertebrate community structure, affecting food web diversity. This may in turn limit foraging opportunities for HCP species exposed to this stressor, affecting survival, growth, and fitness.

The effects of altered community structure on HCP species are complex and variable depending on the nature of the changes and how these species interact with the altered environment. From an ecological perspective, alterations in community structure are generally viewed as negative overall, even though effects on individual species can be negative, positive, or neutral. Applying a worst-case scenario perspective, the effects must be viewed as negative because of the potential for adverse effects on survival, growth, and fitness of any native species within the affected environment. Because these effects are effectively permanent or at least long term on the scale of the life of the structure, they are equated with a high risk of take.

9.1.4 Riparian Vegetation Modifications

Dams alter the extent to which riparian vegetation influences temperature in riverine environments. By greatly expanding the surface area, impoundments limit the shading and ambient temperature buffering influence of the riparian zone upstream of the dam. In downstream reaches, alterations in riparian vegetation characteristics and channel morphology caused by the effects of dams can alter the influence of vegetation on stream temperatures, allochthonous inputs to the riverine ecosystem, and the influence of riparian vegetation on habitat complexity.

9.1.4.1 Altered Shading, Solar Input, and Ambient Air Temperature

Loss of riparian shading can demonstrably affect the temperature of streams and lower order river environments, producing a range of potential effects on fish and wildlife species. In higher order river environments, this effect is far less pronounced. Water temperatures in systems of this nature are less influenced by localized shading and ambient air temperature than by the combined effects of basin conditions in upstream areas. Temperature conditions downstream of the dam will be strongly influenced by the impoundment, as well as how the dam is built and operated. For example, dams that spill water from surface layers of the impoundment during summer months when the impoundment is stratified may cause significant increases in downstream temperatures. (The influence of impoundments on water temperatures is discussed in Section 9.1.6 (*Water Quality Modifications*). In contrast, dams that release flows drawn from deeper, cold water layers of the reservoir may create downstream temperatures that are significantly cooler than the natural temperature range. In smaller rivers and streams, the effects of dams on ecosystem connectivity and channel morphology compound the influence of the impoundment on temperatures. These effects may lead to fragmentation of the river–floodplain connectivity, decreasing the influence of stream shading and altered ambient temperatures in downstream reaches.

On this basis, the risk of take associated with this impact mechanism is viewed as variable, depending on the nature of the project and the type of environment in which it is implemented. Using the worst-case scenario perspective, the effects of altered stream temperatures must be equated with a high risk of take due to the long-term nature of the habitat alteration and the potential effects on survival, growth, and fitness of HCP species.

9.1.4.2 Altered Bank and Shoreline Stability

Removal of riparian vegetation can affect shoreline stability through the reduction in root cohesion and the loss of LWD inputs that affect localized erosion and scour conditions. Dam projects may cause intermediate-term alteration of riparian conditions in downstream reaches, because hydraulic and geomorphic changes may lead to fragmentation of riverine and floodplain habitat. Once riparian vegetation encroachment is established adjacent to the modified channel bank, instability is likely to decrease, unless downcutting caused by sediment starvation leads to long-term instability addressed in Section 9.1.2.1 (*Altered Flow Regime, Channel Geometry, and Substrate Composition*). The risk of take associated with this stressor varies depending on species-specific sensitivity to increased turbidity. In general, more motile fish species experience only temporary behavioral alteration and a low risk of take. In contrast, less motile fish life-history stages or sessile invertebrates could experience a moderate to high risk of take from decreased survival due to substrate sedimentation and smothering, as well as decreased growth and fitness due to the effects of high turbidity on foraging success.

9.1.4.3 Altered Allochthonous Inputs

Riparian vegetation is an important source of nutrient input to the aquatic environment, strongly influencing the productivity of the aquatic food chain. Allochthonous nutrient inputs include sources such as insect-fall, leaf litter and other organic debris, and LWD inputs that contribute both organic material and habitat complexity. The importance of allochthonous inputs to riverine food web productivity decreases along a downstream gradient. However, as rivers grow in size, the contributions of autochthonous production and nutrient cycling to the food web increase. As noted in Section 9.1.3.1 (*Altered Longitudinal Connectivity*), dams alter the transport and cycling of autochthonous nutrients by fragmenting longitudinal connectivity. This has been shown to affect food web productivity in downstream reaches.

Therefore, the magnitude of this impact mechanism varies depending on the extent to which a dam project affects downstream riparian conditions, which in turn affects allochthonous inputs. In lower order streams, allochthonous inputs are more important to food web productivity, while they provide a minor contribution in the lower reaches of large river systems. On this basis, the loss of allochthonous production from a dam project near the mouth of a large river will produce related stressors of potentially lower magnitude than a dam on a small, higher elevation stream. In such cases, a localized reduction in food web productivity might result, leading to decreased foraging opportunities, decreased overall habitat suitability, and decreased growth and fitness. This equates to a moderate risk of take for a range of HCP species that are dependent on riverine rearing conditions.

9.1.4.4 Altered Habitat Complexity

The influence of riparian vegetation on riverine habitat complexity is broadly recognized. Modification or loss of riparian vegetation alters habitat complexity in a number of ways, primarily through the loss of undercut banks, root structure, and LWD inputs to the channel. The hydraulic and geomorphic effects of riparian vegetation modification can lead to further alterations in habitat complexity. This impact mechanism presents a potential risk of take for a broad range of species dependent on riverine aquatic ecosystems through a variety of species-specific stressors. Depending on the particular life history of the affected species, alteration in habitat complexity may limit the availability of suitable spawning, resting, and rearing habitat, and may alter foraging opportunities and predation exposure. In general, fish species that are dependent on habitats potentially affected through this mechanism of impact by dam development are likely to experience decreased spawning success and/or decreased survival, growth, and fitness due to an overall reduction in suitable habitat area. This equates to a moderate risk of take, which applies broadly across all species exposed to the stressor.

9.1.4.5 Altered Groundwater–Surface Water Interactions

Dams alter the interaction between groundwater and surface water in downstream reaches, leading to potential alteration or degradation of the riparian community. Sufficient modification of riparian vegetation may further influence hyporheic exchange, causing a cascading effect in downstream reaches. Risk of take associated with altered groundwater–surface water interactions is discussed in Section 9.1.2.2 (*Altered Groundwater-Surface Water Interactions*) under *Hydraulic and Geomorphic Modifications* (Section 9.1.2).

9.1.5 Aquatic Vegetation Modifications

Dam projects can extensively modify the aquatic vegetation community through the effects of the structure on hydraulic and geomorphic conditions in riverine ecosystems. Aquatic vegetation is a relatively minor component of the ecological structure of riverine and lacustrine systems in Washington State. Aside from native emergent vegetation confined to a relatively narrow range of depths, a large portion of aquatic vegetation species in rivers and lakes are invasive species.

An obvious immediate effect of dam construction is the alteration or elimination of vegetation in the construction footprint. Once established, the impoundment creates a lentic environment suitable for the establishment of emergent vegetation, as well as a colonization opportunity for invasive species. This is particularly true in larger reservoirs that invite recreational boating, which is a well-known vector for invasive species introductions. The subsequent effects of the dam on hydraulic and geomorphic conditions in downstream reaches can also change habitat suitability for native and invasive plant species, altering aquatic vegetation community structure.

Submechanisms of impact associated with the alteration of aquatic vegetation include changes in autochthonous production and altered habitat complexity. The nature of these mechanisms and related stressors varies between riverine and lacustrine habitats; in most river systems in the Pacific Northwest, aquatic vegetation plays a relatively small ecological role. In contrast, vegetation plays a more significant role in lacustrine habitats, and impoundments create suitable environments for these communities to establish. Dam projects established at natural lake outlets (e.g., Lake Chelan, Chester Morse reservoir) may degrade the aquatic vegetation community by altering natural lake hydrology. Operational water level fluctuations may restrict the range of depths over which vegetation can persist, limiting the extent to which they can contribute autochthonous production and habitat structure.

As with the other categories of impact mechanisms addressed in this white paper, the nature and scale of aquatic vegetation modification are dependent on the size and design of the individual dam project, in combination with operational parameters and site-specific conditions. Therefore, the risk of take associated with this type of project is difficult to assess. Certain species may benefit from the habitat structure and food web complexity provided by expanded aquatic vegetation in impoundment environments, but may be detrimentally affected by the overall conversion of riverine to lacustrine habitat. Other species may realize no benefits, while still realizing the same harm. In general, changes in aquatic community structure are representative of the larger scale effects of dams imposed through hydraulic and geomorphic modifications and ecosystem fragmentation. Because aquatic vegetation modification is so closely related to these effects, the risk of take associated with this impact mechanism is rated as moderate.

9.1.6 Water Quality Modifications

The installation of a dam on a river system will invariably have a significant and pervasive effect on water quality conditions. Dam construction is a large undertaking, involving a number of water quality effects such as increased sedimentation, alteration of pH, and the potential introduction of toxic substances to surface waters. Once in place, the ecological fragmentation imposed by the structure, changes in biogeochemical processes that occur within the impoundment, and the effects of hydraulic and geomorphic modification on downstream reaches can in turn result in a number of changes in water temperature and chemistry. These effects and the related risk of take are described in the following sections.

9.1.6.1 Altered Temperature Regime

Dams result in the long-term alteration of the aquatic temperature regime in riverine systems through a variety of mechanisms. The predominant factor is the conversion of lotic riverine habitats to lentic lacustrine environments exposed to increased insulation. These habitats tend to stratify during summer months, significantly increasing water temperatures. Depending on how dams are constructed and operated, they can also significantly alter downstream temperatures. Release of water from the impoundment can lead to increased or decreased temperatures in downstream habitats relative to natural conditions, depending on whether the water is released from relatively warm surface layers, or cooler water below the stratified surface layer.

In either case, these effects, as mentioned, will persist for the life of the structure and have the potential to affect the survival, growth, and fitness of HCP species. Therefore, this impact mechanism equates to a high risk of take.

9.1.6.2 Altered Dissolved Oxygen

The ecological effects imposed by dams can lead to alterations in the concentration of dissolved oxygen and other gasses in surface waters. Two predominant submechanisms occur: decreased dissolved oxygen concentrations caused by eutrophication in the impoundment and potentially surface waters downstream of the dam, and supersaturation of dissolved gasses (predominantly DO, but also nitrogen). If dissolved oxygen concentrations drop below optimal levels, fish will begin to exhibit stress and avoidance behavior. DO concentrations below tolerance thresholds, or depressed DO in combination with elevated water temperatures, may be sufficient to cause mortality, particularly for less-motile life-history stages that are unable to avoid these effects.

Gas supersaturation occurs downstream of impoundments, and is a function of the extreme turbulence created by spillways and other dam structures. Sufficient exposure to supersaturated conditions has been shown to cause mortality in laboratory conditions, and gas bubble disease, which has been shown to cause injury to juvenile salmonids, is known to occur *in situ*.

Less specific information is available regarding the effects of depressed DO levels on invertebrate HCP species. Mussels are known to be intolerant of low DO levels, while the sensitivity of other species is less certain. Given the predilection of all freshwater mollusk HCP species for flowing water environments, however, it is reasonable to conclude that these species are adapted to environments with relatively high natural DO levels. Therefore, depression of DO levels caused by eutrophication in impoundments would be considered a likely adverse effect.

In summary, both increased and decreased DO levels can lead to adverse effects on survival, growth, and fitness of fish populations exposed to these conditions. The collective effects of dams on dissolved oxygen conditions are caused by conditions in the impoundment, as well as the design and operation of the dam itself. They are, therefore, long-term effects that will last for the lifetime of the structure. Therefore, they must be equated with a high risk of take.

9.1.6.3 Altered Suspended Solids and Turbidity

Bank channel and bed disturbance, equipment use, and rewatering of work areas during dam construction are likely to result in a short-term increase in suspended sediment loading to riverine environments downstream of the structure. The subsequent geomorphic effects of the structure on these environments may lead to bank and channel bed erosion that may cause chronic elevation in suspended sediment load as the channel adjusts to the new hydraulic and hydrologic regime imposed by the hydromodification. The effects of elevated suspended sediments vary depending on the magnitude of the stressor and the sensitivity of the species or life-history stage exposed to the stressor.

Non-motile species or life-history stages exposed to pulses of high concentrations of suspended sediments may suffer direct mortality, injury, or extreme physiological stress, while motile species may be able to avoid these stressors. Stressors of this magnitude would typically be expected during the construction phase and would occur most likely as short-term construction-related impacts. Chronic elevation in suspended sediment levels caused by channel adjustments in downstream reaches would be less likely to reach levels sufficient to cause direct mortality but may affect growth and fitness over the intermediate to long term.

In contrast, dams can lead to a reduction in natural suspended sediment loading downstream of the structure. Impoundments provide a low-energy environment that encourages settling of fine sediments transported from upstream areas of the watershed. This may lead to a decrease in suspended sediment levels in downstream reaches. In contrast, eutrophication may encourage elevated turbidity levels in the impoundment, which would be transported to downstream reaches.

On balance, the long-term risk of take from changes in suspended sediments and turbidity caused by dams will be variable depending on site-specific conditions. However, given the potential for short-term injury or mortality resulting from elevated suspended sediment levels associated with construction, a high risk of take must be assumed for this submechanism for HCP species that occur in suitable riverine and lacustrine environments.

9.1.6.4 Increases in Contaminated Sediment

Due to their tendency to capture fine sediments and the tendency of certain contaminants to sorb to small organic and inorganic particles, dams may provide a mechanism for the accumulation of contaminated sediments within the impoundment. In general, these sediments are sequestered and typically become capped as new layers of sediment recruitment are deposited in the impoundment. However, these sediments may be released into the environment during maintenance dredging, or during eventual dam removal. This could result in the release of large volumes of contaminated material over a relatively short period of time, in combination with high levels of suspended sediments overall. Beyond the effects of suspended sediment loading, exposure to toxic substances in contaminated sediments can lead to effects on the survival, growth, and fitness of exposed species. These effects would be expected to be short term and acute in duration and are therefore equated with a moderate risk of take.

9.1.6.5 Altered pH Levels

Dams can lead to the alteration of pH levels through two primary mechanisms of impact: construction, and changes in community structure caused by ecosystem fragmentation. Dams are often constructed of concrete, which is caustic when curing. Concrete leachate released to surface waters from runoff or curing surfaces "in the wet" can increase pH levels well beyond levels capable of causing injury or mortality in any HCP species. This effect is typically short-term in nature and moderates as the concrete cures. If adequate procedures are not in place to protect against this water quality impact, this effect is equated with a high risk of take with potential exposure over a short-term period.

In contrast, impoundments created by dams can create conditions for long-term alteration of pH levels within the impoundment. As noted in Section 9.1.3 (*Ecosystem Fragmentation*) and Section 9.1.5 (*Aquatic Vegetation Modifications*), impoundments can create conditions favorable for the growth of aquatic vegetation and alter the transport and processing of nutrients, potentially to the point where eutrophication can occur. These complex changes in environmental structure can significantly alter pH levels within the impoundment. CO_2 combines with water in solution to form carbonic acid, which measurably decreases pH. Photosynthesis by aquatic vegetation and phytoplankton leads to decreased CO_2 and increased DO during daylight hours, while respiration causes the opposite effect after dark. In eutrophic systems, phytoplankton blooms and subsequent die-offs of aquatic vegetation and plankton can cause a rapid spike in respiration, which rapidly depletes DO levels and increases CO_2. These changes can lead to pH fluctuations within the impoundment environment that may exceed effects thresholds for certain HCP species. In combination with depleted DO, elevated temperatures, and other water quality effects imposed by impoundments, this stressor could cause behavioral avoidance, increased stress and physiological injury, or even mortality to HCP species adapted to cold water and high DO environments with relatively stable pH conditions. In certain impoundment environments, altered pH conditions could occur chronically on a seasonal or annual basis over the life of the structure, and could be limiting to the survival, growth, fitness, and/or spawning productivity of HCP species living within or migrating through the affected environment. Therefore, these effects would be equated with a high risk of take.

9.1.6.6 Altered Nutrient Loading

Impoundments alter the transport and cycling of nutrients within riverine systems. This submechanism in turn supports changes in community structure and aquatic vegetation imposed by ecosystem fragmentation, and drives the water quality effects caused by eutrophication (altered DO and pH levels). The risk of take associated with these submechanisms is discussed in Sections 9.1.3.5 (*Altered Community Composition*), 9.1.5 (*Aquatic Vegetation Modifications*), 9.1.6.2 (*Altered Dissolved Oxygen*), and 9.1.6.5 (*Altered pH Levels*).

9.1.6.7 Introduction of Toxic Substances

Dam projects present multiple pathways for the introduction of a range of toxic substances to the aquatic environment, primarily through construction activities and, in some cases, the use of treated wood materials in the structure. Dams may also indirectly encourage pollutant and nutrient loading by supporting the development of additional infrastructure and expanded recreational vessel use in the impoundment. Depending on the nature and concentration of the contaminant, toxic substance exposure can cause a range of adverse effects in exposed species. In extreme cases, these effects can include direct mortality (e.g., exposure of immobile lamprey ammocoetes buried in bottom substrates, fish exposed to accidental vessel spills in enclosed embayments). More commonly, chronic, low-level exposure to a variety of contaminants is likely to cause physiological injury and/or contaminant bioaccumulation, leading to decreased growth and fitness. This presents a moderate risk of take to species potentially exposed to this stressor.

9.2 Weirs

Weirs include both temporary and permanent structures constructed to control the movement of water, sediments, or organisms in riverine and floodplain environments. For the purpose of this white paper, weirs are assumed to create impoundments or divert streamflow and act similar to a dam. Weir structures created for fish passage and habitat restoration purposes are not included here; these types of structures are addressed in other white papers (Herrera 2007a, 2007e, respectively). The risk of take analysis for weirs focuses on the worst-case scenario design for this subactivity type. Weirs of this type are permanent, typically concrete structures that span the entire channel and create a barrier to fish passage.

Species-specific risk of take ratings for weir development and operation are presented by impact mechanism in Table 9-2. The specific stressors and related risk of take from impact mechanisms caused by this subactivity type are described in the following subsections.

9.2.1 Construction and Maintenance

Construction and maintenance related submechanisms of impact, stressors, and related risk of take associated with weir development are similar to those discussed for dams (Section 9.1) in Section 9.1.1 (*Construction and Maintenance)).* Applying a worst-case scenario perspective, the largest weirs may be comparable in scale to smaller dams, implying that the construction-related impacts would also be similar.

9.2.2 Hydraulic and Geomorphic Modifications

With the exception of altered flow regime, the submechanisms of impact, stressors, and related risk of take from hydraulic and geomorphic modifications associated with weir development are similar to those discussed in Section 9.1.2 (*Hydraulic and Geomorphic Medications*) under

Section 9.1 (*Dams)*. The effects of the remaining submechanisms (channel geometry, substrate composition, and stability) and related risk of take are also otherwise similar.

9.2.3 Ecosystem Fragmentation

The submechanisms of impact, stressors, and related risk of take from ecosystem fragmentation associated with weir development are similar to those discussed in Section 9.1.3 (*Ecosystem Fragmentation*) under Section 9.1 (*Dams)*, but to a lesser degree. Because weirs are not intended to create impoundments, the fragmentation of longitudinal connectivity associated with these structures is restricted to effects on the passage of fish and other organisms, as well as the downstream transport of LWD and organic material. Similarly, there is a lesser effect on community composition. The effects of the remaining submechanisms (altered longitudinal connectivity, altered river-flood plain connectivity, altered LWD transport, altered groundwater-surface water interactions) and related risk of take are otherwise similar.

9.2.4 Riparian Vegetation Modifications

The submechanisms of impact, stressors, and related risk of take from riparian vegetation modifications associated with weir development are similar to those described in Section 9.1.4 (*Riparian Vegetation Modifications*) under Section 9.1 (*Dams*), but occur to a lesser degree. Because weirs are not intended to create impoundments, the effects on riparian vegetation upstream of the structure are expected to be relatively limited (with the exception of the permanent alteration of riparian vegetation in the structural footprint). Downstream impacts are predominantly driven by the hydraulic and geomorphic effects of the structure, particularly on hyporheic exchange, as described for dams. Accordingly, the effects of weirs imposed through altered stream shading and temperature regime, altered allochthonous inputs, altered habitat complexity, and altered groundwater-surface water interactions are similar, but these effects occur to a smaller degree than they do in environments modified by dams because the scale of weir projects is typically smaller.

9.2.5 Aquatic Vegetation Modifications

Weirs may lead to modification or loss of aquatic vegetation in the project footprint and within the zone of hydraulic and geomorphic effects imposed by the structure. However, because the role that aquatic vegetation plays in most riverine environments in Washington State is relatively limited, the degree to which weirs would alter the aquatic vegetation community is expected to be limited. Moreover, once the channel has adjusted to the presence of the weir, the aquatic vegetation community would be expected to recover to some extent. Given the anticipated degree and longevity of these effects, the risk of take resulting from the submechanisms of impact associated with aquatic vegetation modification (altered autochthonous production and altered habitat complexity) is expected to be low to moderate depending on the species-specific sensitivity to these impacts.

9.2.6 Water Quality Modifications

Sources of water quality modification resulting from weir development are associated primarily with project construction and include increases in suspended sediments and turbidity, altered pH levels, and the introduction of toxic substances. The risk of take associated with these submechanisms of impact is similar to that described for dams in Sections 9.1.6.3 (*Altered Suspended Sediments and Turbidity*), 9.1.6.5 (*Altered pH Levels*), and 9.1.6.7 (*Introduction of Toxic Substances*), respectively.

9.3 Dikes and Levees

Dikes and levees are extensive hydromodifications designed to prevent flooding in low-lying landscapes, and to protect and promote human uses. By preventing regular tidal or floodwater inundation, these structures facilitate the conversion of wetland, floodplain, or estuarine habitats for terrestrial uses such as agriculture and development. Extensive in size and pervasive in effect, this subactivity type imposes a number of ecological stressors on the environment through essentially permanent alteration of habitat and water quality conditions. HCP species occurring in environments modified by these types of structures will typically experience a high risk of take from one or more impact mechanisms.

Species-specific risk of take ratings for dike and levee development and maintenance are presented by impact mechanism in Table 9-3. The specific stressors and related risk of take from impact mechanisms caused by this subactivity type are described in the following subsections.

9.3.1 Construction and Maintenance

The construction of dikes and levees represents a significant construction effort, involving the use of heavy machinery, the introduction of extensive fill, and the removal of riparian vegetation throughout the length of the project. Maintenance of these structures includes similar activities, operating at a lesser magnitude and scale at an annual to decadal frequency. These submechanisms are capable of imposing a variety of short-term stressors on the aquatic environment.

9.3.1.1 Equipment Operation and Materials Placement

The operation of heavy construction equipment to build or maintain dikes and levees imposes stressors in the form of physical and visual disturbance of bank and channel habitat, and, potentially, increased underwater noise from in-water equipment use and materials placement. The magnitude of these stressors will vary widely, depending on the scale of the project in question and the specific construction measures used. Applying a worst-case-scenario perspective, the magnitude of these stressors can be significant.

The literature on underwater noise levels produced by heavy equipment use is quite limited, and this subject is considered a data gap. In general, however, noise produced by heavy equipment during the in-water operation of heavy equipment is unlikely to exceed established injury thresholds but may well exceed disturbance thresholds. Therefore, projects involving some element of in-water work are expected to result in a moderate risk of take from underwater noise. Visual disturbance during construction would have similar effects. Physical disturbance is similarly expected to produce only a moderate risk of take due to temporary disturbance and displacement for most species exposed to this stressor. In contrast, sessile or otherwise non-motile species or life-history stages are an exception, as they will be unable to escape or avoid physical disturbance. Therefore, they are at increased risk of mechanical injury from crushing or burial during construction, which constitutes a high risk of take.

9.3.1.2 Bank/Channel/Shoreline Disturbance

Bank, channel, and/or shoreline disturbance during the construction and maintenance of dikes and levees by definition involves significant disturbance and alteration of stream banks and lacustrine and marine shorelines. This disturbance causes short-term water quality impacts, as well as long-term (essentially permanent) modification of hydraulic and geomorphic conditions and ecosystem connectivity. The short-term water quality effects of channel and bed disturbance may lead to behavioral and physiological stress on species or life-history stages exposed to the disturbance, or may limit the availability and suitability of habitats for sensitive life-history stages during critical periods. Non-motile species exposed to these stressors may face immediate effects on survival if occupied habitats are eliminated, or may experience injury or mortality from related water quality effects. These effects would be equated with a moderate to high risk of take, depending on species-specific sensitivity.

9.3.1.3 Dewatering, Flow Bypass, and Fish Handling

The effects of temporary dewatering and flow bypass during construction and maintenance of dikes and levees are similar to those described for dams in Section 9.1.1.4 (*Dewatering, Flow Bypass, and Fish Handling*). This submechanism is equated with a high risk of take.

9.3.2 Hydraulic and Geomorphic Modifications

Dikes and levees may represent a significant alteration of the aquatic environment. Many of the habitat effects of this subactivity type manifest through the modification of hydraulic and geomorphic processes in the affected environment. These effects are significant and become effectively permanent, given the longevity of these structures and the tendency for valuable property improvements and infrastructure to develop landward of them. Risk of take associated with this impact mechanism is described in the following sections.

9.3.2.1 Altered Flow Regime

Hydromodifications in the form of dikes and levees alter flow conditions in riverine environments by fragmenting the channel from floodplain habitats. By preventing the flooding of adjacent terrestrial and riparian habitats, high flows are concentrated in the stream channel, accelerating flow velocity and erosive forces. Reduced floodplain storage of water in hydromodified areas may induce flooding in reaches upstream and downstream of the structure in areas where flooding otherwise would not occur. The effects of this stressor on HCP species are complex and variable, depending on the position of the hydromodification in the riverine environment and how the affected habitats are used by HCP species. Applying a worst-case scenario perspective, these pervasive, long-term effects would be expected to reduce habitat suitability for species utilizing the affected environment, limiting individual survival, growth, and fitness and overall population productivity. This equates to a high risk of take.

9.3.2.2 Altered Channel Geometry, Altered Substrate Transport

Dikes and levees unavoidably lead to changes in channel geometry and substrate composition and stability, either by design or through effect. These structures are often created in conjunction with channel straightening and simplification to accelerate the flow of water through the landscape, to facilitate the conversion of this land to human uses. Channel geometry typically becomes more simplified, and substrate composition and stability are altered through the loss of sources of sediment recruitment and altered sediment transport capacity imposed by channel simplification and alteration of the flow regime.

Channel geometry and substrate composition and stability are dominant factors determining aquatic habitat structure in riverine environments. Alteration of any of these habitat components can change the suitability of the habitat for various life-history stages of HCP species. These habitat alterations are essentially permanent and continuous, and can lead to changes in the productivity of the habitat for spawning, forage, rearing, and refuge. In a worst-case scenario, these effects are in turn likely to lead to reduced spawning success as well as reduced survival, growth, and fitness for species and life-history stages dependent on the affected habitat. This equates to a high risk of take for species with exposure to these impact mechanisms.

These effects are not as pronounced in marine and lacustrine systems. The predominant effects of dikes and levees on these environment types are imposed through ecosystem fragmentation, addressed in the following section.

9.3.3 Ecosystem Fragmentation

An unintended consequence of dikes and levees is the fragmentation of ecological connectivity between the aquatic and terrestrial environments. By aiding the conversion of low-lying floodplain and wetland habitats to terrestrial uses, these structures sharpen the gradient between the aquatic and terrestrial landscape, imposing a number of ecological stressors on organisms adapted to the affected environments. The submechanisms of impact, related ecological stressors, and risk of take associated with ecosystem fragmentation operate in a slightly different fashion between riverine and lacustrine/marine habitats. A discussion of the risk of take occurring in these environment types is provided in the following sections.

9.3.3.1 Riverine Environments

9.3.3.1.1 Altered Longitudinal Connectivity

Hydromodification of riverine environments by dikes and levees reduces the structural complexity of instream habitat by changing the channel geometry and influencing the recruitment, transport, and retention of sediments and LWD. Complex channels capture and retain sediment, which promotes the formation of pools and other hydraulically complex features. This hydraulic complexity in turn encourages the sorting and deposition of sediments and organic material in diverse patches, supporting food web productivity and providing spawning and rearing habitat for a diverse array of species. This diversity of habitat patches supports a biologically diverse community. Hydromodification promotes simplification of the riverine environment, reducing the longitudinal distribution and frequency of these habitat patches across the riverine landscape. This reduction in habitat complexity leads to reduced food web productivity, as well as the reduced availability of habitats suitable for HCP species that occur in these environments. Because these effects are extensive and effectively permanent, this impact mechanisms equates to a high risk of take for HCP species.

9.3.3.1.2 Altered River–Floodplain Connectivity

Dikes and levees purposefully disconnect of floodplain and off-channel habitats from the riverine ecosystem. This form of ecosystem fragmentation limits the extent to which river flows interact with the floodplain and terrestrial riparian ecosystem, disconnecting the stream channel from important sources and sinks of organic matter, nutrients, and pollutants. This in turn may limit food web productivity, affecting the survival, growth, and fitness of any species dependent on the riverine environment for rearing. In addition, this loss of connectivity may limit the availability of important habitat types for HCP species. For example, side channel habitats are preferentially selected by various species of salmonids (e.g., sockeye salmon) for spawning. These habitats also provide key winter rearing and storm refuge habitats for coho salmon, steelhead, spring Chinook, native char (bull trout and Dolly Varden), and other species. Floodplain wetlands are also highly productive refuge habitats for a variety of species, such as coho salmon, during high winter flows. The reduction in suitable refuge and foraging habitat area caused by ecosystem fragmentation increases competition for remaining habitat, predation risk, and risk of displacement to habitats unfavorable for rearing. Collectively, these long-term ecological stressors pose a high risk of take for HCP species that occur in the affected riverine environment.

9.3.3.1.3 Altered Groundwater–Surface Water Interactions

Modification of hydraulic, geomorphic, and riparian conditions caused by dikes and levees can influence and alter groundwater and surface water exchange in the project area and downstream. This hyporheic exchange is an important component of ecosystem function (including water quality moderation) in riverine environments. Therefore, this impact mechanism has the potential to affect juvenile and/or adult survival, growth, and fitness, and in some cases the spawning productivity of a range of species. Because this effect will be pervasive and essentially permanent, this mechanism is generally equated with a moderate to high risk of take for species exposed to this stressor, depending on species-specific life-history characteristics. Species with a high risk of take include those with life-history stages that are dependent on hyporheic exchange for its beneficial effects on water temperature and dissolved oxygen levels. For example, most salmonids preferentially spawn in areas with groundwater-induced upwelling, which promotes oxygenation of spawning gravels. Alteration of hyporheic exchange in environments suitable for spawning could potentially affect egg survival and reduce the availability of suitable spawning habitat, resulting in reduced spawning success. Similarly, groundwater inflow can provide important thermal refugia for migrating adult and rearing juvenile salmonids during periods with high water temperatures. A reduction in the amount of thermal refugia may negatively affect survival during these life-history stages. Similar effects would be expected for other coldwater fish species with low thermal tolerance thresholds, such as pygmy whitefish. More generally, hyporheic exchange plays a key role in nutrient cycling and food web productivity in alluvial bed rivers. Projects resulting in significant alteration of hyporheic exchange could adversely affect food web productivity, limiting foraging opportunities for fish and invertebrate species dependent on these types of environments.

9.3.3.2 Lacustrine Environments

Dikes and levees could potentially be implemented in lacustrine environments, specifically in river deltas or other adjacent low-lying areas converted to agricultural land. These projects intentionally fragment lacustrine and floodplain habitats, preventing access to these habitats and facilitating their conversion for terrestrial uses. This imposes effects through a number of submechanisms and related stressors. The risk of take resulting from these impact mechanisms is strongly linked to species-specific dependence on lacustrine environments.

By design, dikes and levees prevent inundation of adjacent lowland habitats during high-water periods. Habitats in the physical footprint of the structure are permanently lost as a result of construction and, by design, the structure prevents access to what may be productive wetland habitat. This facilitates the conversion of this habitat for terrestrial uses. Hydraulic and geomorphic effects waterward of the structure can alter bathymetry and current and circulation patterns within the nearshore environment, further altering habitat conditions and potentially altering desirable habitat types.

On balance, dikes and levees constructed in lacustrine environments for human uses permanently fragment and alter habitats shoreward of the structure, and may render productive habitats waterward of the structure less suitable for species dependent on estuarine environments. Collectively, habitat fragmentation caused by dikes and levees in the lacustrine environment would be expected to affect the survival, growth, and fitness of affected species, as well as the overall population productivity of HCP species dependent on nearshore lacustrine habitats. These effects are associated with a high risk of take because they are essentially permanent.

9.3.3.3 Marine Environments

Dikes and levees could potentially be implemented in the marine environment, specifically in estuarine areas on river deltas converted to agricultural land. These projects intentionally interrupt the tidal inundation of estuarine habitats, preventing access to these habitats and facilitating their conversion for terrestrial uses. This imposes effects through a number of submechanisms and related stressors. The risk of take resulting from these impact mechanisms is strongly linked to species-specific dependence on the nearshore and estuarine environment.

By design, dikes and levees prevent tidal inundation of lowland habitats adjacent to marine environments. Habitats in the physical footprint of the structure are permanently lost as a result of construction and, by design, the structure prevents access to what may be productive habitat. This facilitates the conversion of this habitat for terrestrial uses. Hydraulic and geomorphic effects seaward of the structure can alter bathymetry, salinity, tidal exchange, and circulation patterns within the estuarine and nearshore environment, further altering habitat conditions and potentially altering desirable habitat types.

On balance, dikes and levees constructed in marine environments for human uses permanently fragment and alter habitats shoreward of the structure, and may render productive habitats seaward of the structure less suitable for species dependent on estuarine environments. In the case of organisms with a planktonic life-history stage, the effects of dikes and levees may limit the dispersal and retention of eggs and larvae to areas suitable for rearing. Collectively, habitat fragmentation caused by dikes and levees in the marine environment would be expected to affect the survival, growth, and fitness of affected species, as well as overall population productivity. These effects are associated with a high risk of take because they are essentially permanent.

9.3.4 Riparian Vegetation Modifications

Dike and levee development can result in extensive modification of riparian vegetation along a significant length of stream channel, or marine or lacustrine shoreline. These effects are often particularly pronounced in riverine environments, by virtue of channel simplification and the fragmentation of floodplain habitats that accompanies these structures. Riparian vegetation is often removed to create dikes and levees. Once the structures are established, vegetation is often managed to prevent the degradation of structural integrity caused by root penetration. The risk of take for stressors resulting from this mechanism is discussed in the following sections by environment type.

9.3.4.1 Riverine Habitats

Modification of riverine riparian vegetation to accommodate dikes and levees is often extensive. Removal occurs through both the direct removal of vegetation from the stream bank in the footprint of the structure, and channel realignment and fragmentation of floodplain habitats that reduce the length of channel exposed to the riparian zone. Once the structures are established, they are often managed to restrict the growth of larger trees and shrubs. The reduction in riparian vegetation and potential riparian area imposes a variety of stressors through the following submechanisms.

9.3.4.1.1 Altered Shading, Solar Input, and Altered Ambient Air Temperature

Loss of riparian shading can demonstrably affect the temperature of streams and lower order riverine environments, producing a range of potential effects on fish and wildlife species. In higher order riverine environments, this effect is far less pronounced. Water temperatures in systems of this nature are less influenced by localized shading and ambient air temperature than by the combined effects of basin conditions in upstream areas. In smaller rivers and streams, the removal of vegetation to accommodate dikes and levees and the effects of these structures on ecosystem connectivity and channel morphology compound the potential temperature effects caused by a loss of shading and ambient temperature regulation. Through these pathways, dikes and levees can lead to unfavorable alterations in stream temperature conditions for HCP species adapted to the unaltered river environment.

Using the worst-case scenario perspective, the effects of altered stream temperatures must be equated with a high risk of take due to the long-term nature of the habitat alteration and the potential effects on survival, growth, and fitness of HCP species.

9.3.4.1.2 Altered Bank Stability

While removal of riparian vegetation can lead to bank instability, hardened dike and levee structures are specifically intended to increase bank stability and to prevent erosion. The level of bank stability provided by the structures is undesirable from an ecological perspective because it prevents natural channel migration processes and limits the recruitment of sediment. The risk of take associated with these undesirable effects is discussed in Section 9.3.2 (*Hydraulic and Geomorphic Modifications).*

9.3.4.1.3 Altered Allochthonous Inputs

Riparian vegetation is an important source of nutrient input to the aquatic environment, strongly influencing the productivity of the aquatic food chain. Allochthonous nutrient inputs include sources such as insect-fall, leaf litter and other organic debris, and LWD inputs that contribute both organic material and habitat complexity. The importance of allochthonous inputs to riverine food web productivity decreases along a downstream gradient. However, as rivers grow in size, the contributions of autochthonous production and nutrient cycling to the food web increase. Dikes and levees alter the transport and cycling of autochthonous nutrients through simplification of channel structure, fragmenting longitudinal connectivity. This has been shown to affect food web productivity in downstream reaches.

The magnitude of this impact mechanism varies depending on the amount of riparian vegetation modified or disconnected from the channel by the dike/levee project, as well as the effects of channel simplification on nutrient cycling. In lower order streams, allochthonous inputs are more important to food web productivity; in the lower reaches of large river systems, they provide a minor contribution. On this basis, the loss of allochthonous production from a dike/levee project near the mouth of a large river will produce related stressors of potentially lower magnitude than an extensive dike/levee system in smaller rivers, or higher in the watershed of larger systems. In such cases, a localized reduction in food web productivity might result, leading to decreased foraging opportunities, decreased overall habitat suitability, and decreased growth and fitness. Due to the long-term nature of these effects, this equates to a high risk of take for a range of HCP species that are dependent on riverine rearing conditions.

9.3.4.1.4 Altered Habitat Complexity

The influence of riparian vegetation on riverine habitat complexity is broadly recognized. Modification of riparian vegetation alters habitat complexity in a number of ways, primarily through the loss of undercut banks, root structure, and LWD inputs to the channel. The hydraulic and geomorphic effects of riparian vegetation modification can lead to further alterations in habitat complexity. This impact mechanism presents a potential risk of take for a broad range of species dependent on riverine aquatic ecosystems through a variety of species-specific stressors. Depending on the particular life history of the affected species, alterations in habitat complexity may limit the availability of suitable spawning, resting, and rearing habitat, and may alter foraging opportunities and predation exposure. In general, fish species that are dependent on habitats potentially affected through this mechanism of impact by dike and levee development are likely to experience decreased spawning success and/or decreased survival, growth, and fitness due to an overall reduction in suitable habitat area. Due to the long-term nature of this submechanism, this equates to a high risk of take, which applies broadly to all HCP species that utilize riparian habitats.

9.3.4.1.5 Altered Groundwater–Surface Water Interactions

Sufficient modification of riparian vegetation may further influence hyporheic exchange, causing a cascading effect in downstream reaches. Risk of take associated with altered groundwater–surface water interactions for dikes/levees is similar to that discussed for dams in Section 9.1.2.2 (*Altered Groundwater-Surface Water Interactions*) under *Hydraulic and Geomorphic Modifications* (Section 9.1.2).

9.3.4.2 Marine and Lacustrine Habitats

Dikes and levees in marine and lacustrine environments are typically created in estuaries and adjacent to low-lying wetlands to accommodate conversion of these lands to terrestrial uses. Riparian vegetation in these environments may include trees and shrubs, but more typically includes fringing estuarine or wetland vegetation.

9.3.4.2.1 Altered Shoreline Stability

As in riverine environments, the removal of shoreline vegetation can lead to bank instability. Hardened dike and levee structures are specifically intended to increase bank stability and prevent erosion. The level of bank stability provided by the structures is undesirable from an ecological perspective because it may lead to unfavorable changes in bathymetric profile, longshore sediment transport, and other important ecological characteristics. The risk of take associated with these undesirable effects is discussed in Section 9.3.2 (*Hydraulic and Geomorphic Modifications).*

9.3.4.2.2 Altered Allochthonous Inputs

Allochthonous inputs to the nearshore environment from marine riparian vegetation include leaf litter and terrestrial insect-fall, as well as inputs of LWD. These inputs clearly contribute to aquatic food web productivity, but the science regarding the significance of these inputs is relatively limited. LWD recruitment is an important contributor to habitat structure. In contrast, allochthonous inputs from saltmarsh or emergent wetland environments may be less extensive. Regardless of magnitude, this stressor has the potential to alter food web productivity and habitat complexity; it is likely to affect the survival, growth, and fitness of those HCP species dependent on the nearshore environment for foraging and rearing during some portion of their life history. This equates to a high risk of take for species with demonstrable dependence on these habitats because these effects will be long term in duration.

9.3.4.2.3 Altered Habitat Complexity

The physical structure provided by marine riparian and lacustrine riparian vegetation, allochthonous inputs of LWD and organic material, and effects on localized microhabitat conditions all contribute to habitat complexity of the nearshore environment. Alteration of habitat complexity can have demonstrable effects on the productivity of those aquatic species dependent on the nearshore environment, particularly fish species that spawn and rear in these areas, through effects on survival, growth, fitness, and spawning productivity. Because the causes of these modifications are long-term in duration, these effects equate to a high risk of take for species with demonstrable dependence on these habitats.

9.3.5 Aquatic Vegetation Modifications

Dikes and levees modify the aquatic vegetation community, resulting in the imposition of potential stressors. The resulting risk of take associated with these stressors varies based on the sensitivity of the HCP species and the environment type in which the stressor exposure occurs.

9.3.5.1 Riverine and Lacustrine Environments

Aquatic vegetation is a relatively minor component of the ecological structure of riverine and lacustrine systems in Washington State. Aside from native emergent vegetation confined to a relatively narrow range of depths, most aquatic vegetation species in rivers and lakes are invasive species. Thus, the risk of take resulting from this impact mechanism is relatively minor in comparison to that occurring in the marine environment.

9.3.5.1.1 Altered Autochthonous Production

Modification of the submerged aquatic vegetation community in lakes and rivers can lead to decreased primary and secondary productivity, which in turn may affect overall food web productivity. In systems where the aquatic vegetation community is an important component of food web productivity, this can lead to a high risk of take through long-term, indirect effects on foraging success, growth, and fitness of species and life-history stages that depend on forage in the nearshore environment. As noted, however, a high risk of take would only apply to species adapted to habitats with naturally abundant aquatic vegetation. Otherwise, only a moderate risk of take would be expected.

9.3.5.1.2 Altered Habitat Complexity

Submerged aquatic vegetation provides habitat structure in nearshore environments, creating vertical dimension and overhead cover. Alteration of habitat complexity can decrease the availability of suitable rearing habitat for species and life-history stages dependent on the nearshore environment, leading to increased predation risk and increased competition for suitable space, leading to long-term effects on survival, growth, and fitness. This equates to a high risk of take for species dependent on aquatic vegetation functions in these environments. As noted, however, a high risk of take would only apply to species adapted to habitats with naturally abundant aquatic vegetation. Otherwise, only a moderate risk of take would be expected.

9.3.5.2 Marine Environments

Submerged aquatic vegetation (including eelgrass, kelp, and other forms of marine algae) and emergent saltmarsh vegetation is an important component of estuarine ecosystems where dike and levee projects are likely to occur. Vegetation provides habitat structure and supports food web productivity relied upon by many species during critical life-history stages in these environment types.

9.3.5.2.1 Altered Autochthonous Production

Autochthonous production by submerged aquatic vegetation is a source of primary and secondary production in the aquatic food web in nearshore marine and estuarine environments. A diversity of species feed directly on live and fragmented submerged aquatic vegetation, forming the basis of the food web for a number of other species. Vegetation in and landward of the structural footprint will typically be destroyed. Alteration of marine and estuarine vegetation may in some cases lead to localized shifts in food web productivity, possibly affecting foraging opportunities for dependent species and life-history stages. This equates to a moderate risk of take resulting from decreased growth and fitness.

9.3.5.2.2 Altered Habitat Complexity

The contribution of submerged aquatic vegetation to habitat structure in nearshore marine environments is well recognized. Numerous species use these habitats for cover and rearing during larval and juvenile life-history stages. Submerged aquatic vegetation also provides spawning habitat for Pacific herring. Alterations of the aquatic vegetation community can reduce the productivity of these habitats for dependent life-history stages, or eliminate the habitat altogether. For example, Newcomb's littorine snail is found only on *Salicornia* spp. (glasswort) in saltmarsh environments. Dike or levee projects that convert saltmarsh environments for terrestrial uses would effectively eliminate the only habitat used by this obligate species. These effects equate to a moderate to high risk of take for HCP species, based on the dependence of the species on nearshore aquatic vegetation and the effectively permanent nature of the habitat modification.

9.3.6 Water Quality Modifications

Sources of water quality modification associated with dikes and levees include increased suspended sediments and the potential introduction of toxic substances during project construction, as well as the effects of riparian and hydraulic and geomorphic modification on stream temperatures. Risk of take associated with these submechanisms is discussed in the following sections.

9.3.6.1 Altered Temperature Regime

Risk of take associated with this submechanism is similar to that discussed for dams in Section 9.1.6.1 (*Altered Temperature Regime*).

9.3.6.2 Altered Suspended Solids and Turbidity

The effects of this construction-related stressor and related risk of take are similar to those described for dams in Section 9.1.6.3 (*Altered Suspended Sediments and Turbidity*). However, the species exposed to the risk of take include both freshwater and marine HCP species that occur in environments potentially suitable for this subactivity type.

9.3.6.3 Introduction of Toxic Substances

Risk of take associated with this submechanism is similar to that discussed for dams in Section 9.1.6.7 (*Introduction of Toxic Substances*).

9.4 Outfalls

Outfalls are structures created to deliver surface water runoff or effluent to aquatic environments. These structures are quite common and are distributed pervasively around the landscape in riverine, lacustrine, and marine environment types. Outfall structures are commonly relatively small in scale and, as such, have relatively limited physical effects on the aquatic environment in comparison to other types of flow control structures. However, outfalls are a significant source of potential take because they facilitate the delivery of nutrients and pollutants to surface waters.

Species-specific risk of take ratings for outfalls are presented by impact mechanism in Table 9-4. The specific stressors and related risk of take from impact mechanisms caused by this subactivity type are described in the following subsections.

9.4.1 Construction and Maintenance

The construction of outfalls typically involves disturbance of bank and shoreline habitat to place the outfall structure and related erosion protection at the outlet. In lacustrine and marine environments, outfall construction may extend through the littoral zone to place the outlet below the water surface, preventing beach erosion. Regardless of configuration, outfall construction involves the use of heavy equipment to place the structure, which is typically composed of precast concrete or metal. Stressors imposed by submechanisms associated with outfall construction and the related risk of take are described in the following sections.

9.4.1.1 Elevated Underwater Noise

The risk of take associated with this submechanism is similar to that described for dams in Section 9.1.1.1 (*Elevated Underwater Noise*), but outfalls operate at a far lesser degree of magnitude. Underwater noise effects would likely be insufficient to cause direct injury, meaning that stressor response would likely be limited to short-term disturbance and behavioral modification. Stressor exposure of this magnitude is equated with a low to moderate risk of take, depending on the size scale of the structure in question.

9.4.1.2 Equipment Operation and Materials Placement

The risk of take associated with this submechanism is similar to that described for dams in Section 9.1.1.2 (*Equipment Operation and Materials Placement*), but because outfalls are much smaller in scale than dams, the effects associated with potential take are similarly smaller in scale and shorter in duration. In a worst-case scenario, outfall construction may involve in-water work, including equipment use and material placement. These activities could result in potential injury or mortality of HCP species having sessile or non-motile life-history stages. These effects are equated with a high risk of take. Motile species or life-history stages would experience temporary disturbance and displacement, potentially affecting survival, growth, and productivity. These effects are equated with a moderate risk of take.

9.4.1.3 Bank/Channel/Shoreline Disturbance

Outfall construction by definition involves bank, channel, or shoreline disturbance. The risk of take associated with this submechanism is similar to that described for dams in Section 9.1.1.3 (*Bank/Channel/Shoreline Disturbance*), but outfalls are smaller in scale and thereby exert a smaller impact. In a worst-case scenario, outfall construction may require significant disturbance of the bank/shoreline and substrate, degrading habitat conditions in the affected habitat and resulting in the release of suspended sediments. These activities could result in potential injury or mortality of HCP species having sessile or non-motile life-history stages. These effects are equated with a high risk of take. Motile adult species or motile life-history stages would experience temporary disturbance and displacement, potentially affecting survival, growth, and productivity. These effects are equated with a moderate risk of take.

9.4.1.4 Dewatering, Flow Bypass, and Fish Handling

Outfall construction may require temporary dewatering and/or flow bypass during construction. The risk of take associated with this submechanism is similar to that described for dams in Section 9.1.1.4 (*Dewatering, Flow Bypass, and Fish Handling)*, but outfalls operate at a lesser degree of magnitude due to the smaller size of the construction footprint. Creation of exclusion areas, fish removal and relocation, and work area dewatering/flow bypass are all activities with the potential to cause injury or mortality to HCP species. Therefore, the effects of this submechanism are equated with a high risk of take.

9.4.2 Hydraulic and Geomorphic Modifications

The degree to which outfalls modify hydraulic and geomorphic conditions in the aquatic environment is a function of the scale of the structure and how it interacts with these physical processes. Outfalls are often configured differently in riverine, marine, and lacustrine environments. Therefore, the risk of take associated with this impact mechanism is addressed separately for each environment type.

9.4.2.1 Riverine Environments

The effects of hydraulic and geomorphic modifications caused by outfalls in marine environments are relatively limited because these structures are typically located on the stream bank and have a relatively small footprint in comparison to other subactivity types considered in this white paper. A broad array of riverine habitat types may be considered suitable for outfall projects. Therefore, effectively all riverine species and life-history stages could be exposed to stressors and experience a resulting risk of take due to hydraulic and geomorphic modification caused by outfalls.

9.4.2.1.1 Altered Channel Geometry, Altered Flow Regime, and Altered Substrate Composition

Channel geometry, flow conditions, and substrate composition are dominant factors determining aquatic habitat structure in riverine environments. Alteration of any of these habitat components can change the suitability of the habitat for various life-history stages of HCP species. Outfalls can alter hydraulic and geomorphic conditions through these submechanisms, but because the size of these structures is typically relatively limited, the magnitude of the effects caused by individual outfall projects is not likely sufficient to affect HCP species survival, growth, and fitness at a large scale. Therefore, the resulting risk of take associated with these effects is likely to be moderate.

9.4.2.1.2 Altered Groundwater–Surface Water Exchange

Hydraulic and geomorphic modifications can influence and alter groundwater and surface water exchange in the vicinity of the modification. This hyporheic exchange is an important component of ecosystem function (including water quality moderation) in riverine environments. Therefore, this impact mechanism has the potential to affect juvenile and/or adult survival, growth, and fitness and, in some cases, the spawning productivity of a range of species. This mechanism is generally equated with a moderate to low risk of take for species exposed to this stressor, depending on species-specific life-history characteristics. Species with a moderate risk of take include those with life-history stages that are dependent on hyporheic exchange for its beneficial effects on water temperature and dissolved oxygen levels.

9.4.2.2 Marine Environments

Outfalls in the marine environment are typically more extensive structurally than those in lacustrine and riverine environments. Marine outfalls typically extend from upland habitats through the littoral zone and discharge into subtidal habitats. Therefore, these projects unavoidably modify hydraulic and geomorphic conditions in the nearshore marine environment, resulting in the imposition of several impact mechanisms and related stressors. The risk of take resulting from these impact mechanisms is strongly linked to species-specific dependence on the nearshore environment.

9.4.2.2.1 Altered Flow Regime (Wave Energy, Current Velocities, and Nearshore Circulation Patterns)

Wave energy, current velocities, and circulation patterns are all important determinants governing nearshore marine habitat characteristics. These factors determine habitat suitability for a number of species-specific life-history processes, and alteration of their characteristics may change habitat suitability. For example, wave energy conditions, currents, and circulation patterns have a strong influence on nearshore water temperatures and on the sorting and transport of sediments. Alteration of substrate conditions may change the suitability of spawning substrate for forage fishes, while alteration of nearshore temperatures may unfavorably change rearing conditions for planktonic larvae.

Marine outfalls are constructed in many different configurations, ranging from structures with energy dissipaters that discharge directly to the beach to buried structures that transect the intertidal zone beneath the substrate and discharge offshore. These latter structures are occasionally exposed by longshore drift and erosion. Outfall structures that are exposed (whether by design or unintentionally) could potentially attenuate wave energy, alter localized circulation patterns, and interrupt longshore sediment transport, leading to changes in beach profile. These changes in ecological characteristics may in turn impose ecological stressors, resulting in adverse effects on HCP species. For example, many marine fish species selectively spawn in locations where current and circulation patterns promote the settling of planktonic larvae in favorable environments for rearing. Alteration of these patterns can cause larvae to be transported to unfavorable environments. Similarly, juvenile fish rearing in nearshore environments selectively choose environments with suitable wave energy and current conditions. Alteration of these characteristics can fundamentally alter habitat suitability for these uses, leading to decreased habitat availability and decreased survival, growth, and fitness. These effects equate to a high risk of take for species that are dependent on these habitats during some phase of their life history due to the long-term life span of this structure type.

9.4.2.2.2 Altered Sediment Supply and Altered Substrate Composition

Sediment supply and substrate composition are fundamental components of the nearshore ecosystem structure. The physical alteration of the shoreline environment caused by large outfalls that are exposed by design or become exposed unintentionally may lead to alterations in sediment supply and substrate conditions through reduced sediment recruitment, as well as the interruption or alteration of longshore sediment transport. In conjunction with altered wave energy, this can lead to changes in substrate conditions that may be beneficial or detrimental to individual species. Because substrate composition is an important determinant of community structure in the nearshore environment, these habitat changes can fundamentally alter community structure and habitat suitability for species dependent on the original habitat condition. This equates to a high risk of take for species that are dependent on these habitats due to long-term effects on the survival, growth, and productivity of exposed life-history stages.

9.4.2.2.3 Altered Freshwater Inputs

Freshwater inputs to the nearshore environment are demonstrably linked to a number of important habitat parameters such as temperatures in forage fish spawning substrates, eelgrass distribution, and habitat selection by certain fish species. Outfalls contribute fresh water to the nearshore marine environment but do so typically with an altered timing and duration of flows, which may not lead to the same desirable habitat characteristics. Moreover, outfall discharges may carry undesirable pollutants leading to degradation of water quality. Due to these factors, the alteration in freshwater inputs imposed by outfalls is viewed to be an ecologically undesirable effect that is long term in duration, potentially leading to reduced survival, growth, and fitness. This equates to a high risk of take for species experiencing stressor exposure.

9.4.2.3 Lacustrine Environments

As with marine environments, outfall projects implemented in the lacustrine environment may unavoidably modify hydraulic and geomorphic conditions, resulting in the imposition of several impact mechanisms and related stressors. The related risk of take is similar to that for marine habitats, determined by the size and scale of the project in question, as well as species-specific dependence on the nearshore lacustrine environment where the stressors associated with these impact mechanisms are manifest.

9.4.2.3.1 Altered Flow Regime (Wave Energy, Current Velocities, and Nearshore Circulation Patterns)

Wave energy, current velocities, and circulation patterns are all important determinants governing nearshore lacustrine habitat characteristics. These processes strongly influence nearshore water temperatures and other water quality parameters, shoreline stability, and the accumulation of allochthonous and autochthonous materials. Outfalls that are exposed to these processes due to either how the structure was initially designed or unintentionally due to erosion may change how they function in the nearshore environment. This in turn may alter the suitability of nearshore habitats for species dependent on these habitats, leading to decreased survival, growth, and fitness. Because outfalls are typically intended to have long operational life spans, the stressors imposed by these effects are likely to be long term in duration. Therefore, the effects of outfalls on wave energy, current, and circulation patterns are equated with a high risk of take for species that are dependent on these habitats during some phase of their life history.

9.4.2.3.2 Altered Sediment Transport and Altered Substrate Composition

Sediment supply and substrate composition are fundamental components of the nearshore ecosystem structure. The physical alteration of the shoreline environment caused by large exposed outfalls may lead to alterations in sediment supply and substrate conditions through reduced sediment recruitment, as well as the interruption or alteration of longshore sediment transport. In conjunction with altered wave energy, this can lead to changes in substrate conditions that may be beneficial or detrimental to individual species. Because substrate composition is an important determinant of community structure in the lacustrine environment, these habitat changes can fundamentally alter community structure and habitat suitability for species dependent on the original habitat condition. Applying a worst-case scenario perspective, an exposed outfall could cause long-term alteration of substrate conditions in the vicinity of the structure. This equates to a high risk of take for species that are dependent on these habitats due to effects on the survival, growth, and productivity of exposed life-history stages given the long-term nature of stressor exposure.

9.4.3 Ecosystem Fragmentation

Ecosystem fragmentation is an impact mechanism that incorporates the collective effects of the loss of habitat within the footprint of the structure, the physical barrier the structure presents in terms of the migration and dispersal of organisms, the transport of LWD and other organic material, and the impact mechanisms imposed by hydraulic and geomorphic modification. The degree to which outfalls cause ecosystem fragmentation is variable, depending on the size and configuration of the structure. This impact mechanism operates differently in riverine versus marine/lacustrine environments due to this factor and the differences in characteristic physical and biological processes that operated in these environment types. Because the effects of outfalls on ecosystem fragmentation are relatively minor in comparison to other flow control structures, the discussion of these effects is limited to the impact mechanism level by environment type.

9.4.3.1 Riverine Environments

The degree to which outfalls cause ecosystem fragmentation in riverine environments is limited. Outfalls in riverine environments are typically located on the bank and discharge at the edge of the stream channel. Therefore, the effects of these structures on longitudinal connectivity are limited. Water quality effects present a possible exception. Concentrated discharge of stormwater or effluents could create a dilution zone with water quality conditions that are sufficiently unfavorable to cause avoidance behavior. If this mixing zone extends across a majority of the channel, it could impose a barrier to fish passage. This would represent fragmentation of longitudinal connectivity. Depending on the duration and frequency of the effect, this could deny access to productive habitats, potentially limiting the survival, growth, fitness, and productivity of affected populations. Under a worst-case scenario, this effect would equate to a high risk of take.

9.4.3.2 Marine and Lacustrine Environments

Outfalls in marine and lacustrine habitats typically take the form of either a pipe with an exposed discharge point on the beach with erosion control, or a buried structure that extends across the littoral zone. Exposed structures on the beach may operate similarly to a barb or other type of perpendicular structure, exerting effects on ecological connectivity through hydraulic and geomorphic modifications. Buried outfall pipes with discharge points located offshore in deep water would not be expected to impose such effects. However, in a worst-case scenario, a buried outfall pipe may become exposed by natural erosional processes, creating a perpendicular barrier across the shoreline and adjacent shallow water habitats. In such cases, the physical presence of the structure and the modification of habitat complexity caused by its effects on hydrologic and geomorphic conditions may fragment habitat connectivity along the shoreline. Many species, such as salmonids, have life-history stages that migrate along shorelines in shallow water environments. Outfalls may force individuals to migrate around the structure, causing behavioral alteration, increased exertion, and potentially increased exposure to predation. Due to the variability of potential effects, the risk of take from ecosystem fragmentation caused by outfalls may range from insignificant (e.g., for buried outfall pipes with discharge points located far offshore) to high (e.g., for exposed outfalls or outfall pipes that create a perpendicular barrier and causing hydraulic and geomorphic modifications of the nearshore environment).

9.4.4 Riparian Vegetation Modifications

The effects of outfalls on riparian vegetation are expected to vary depending on the extent to which the individual structure results in modification of shoreline hydraulic and geomorphic conditions. In general, outfalls would be expected to have a relatively limited effect on riparian vegetation because their onshore construction footprint is relatively small. Once established, the degree to which these structures affect riparian recovery is expected to be similarly limited. However, should the structure impose extensive hydraulic and geomorphic effects that alter bank stability, effects on riparian vegetation could be more extensive. In general, however, outfall structures are not expected to be associated with bank erosion to a degree that would cause widespread losses of riparian vegetation; therefore, effects would be expected to be intermediate-term in nature as riparian vegetation adjusts to changing conditions. On this basis, the risk of take associated with stressors resulting from this impact mechanisms is expected to be moderate.

9.4.5 Aquatic Vegetation Modifications

The effects of outfalls on aquatic vegetation from project construction are expected to be relatively minor given the limited footprint of these structures. Over time, however, these structures may modify the aquatic vegetation through their effects on hydraulic and geomorphic processes, as well as on water quality conditions. These effects and the related risk of take resulting from these structures are described in the following sections.

9.4.5.1 Riverine and Lacustrine Environments

Aquatic vegetation is a relatively minor component of the ecological structure of riverine and lacustrine systems in Washington State. Aside from native emergent vegetation confined to a relatively narrow range of depths, the majority of aquatic vegetation species in rivers and lakes are likely to be invasive species. Thus, the risk of take resulting from this impact mechanism due to the effects of outfalls is relatively minor in comparison to that occurring in the marine environment.

Modification of the submerged aquatic vegetation community in lakes and rivers would typically be limited to the footprint of the structure, and possibly the effects of effluent on vegetation growth. Assuming that effluent concentrations are managed properly, the effects of outfalls on autochthonous productivity and habitat structure would be expected to be minor, and are equated with an insignificant risk of take.

9.4.5.2 Marine Littoral Environments

Submerged aquatic vegetation (including eelgrass, kelp, and other forms of marine algae) is an important component of the marine littoral ecosystem relied upon by many species during critical life-history stages. The autochthonous production by submerged aquatic vegetation is a source of primary and secondary production in the aquatic food web of the marine littoral zone. A diversity of species feed directly on live and fragmented submerged aquatic vegetation, forming the basis of the food web for a number of other species. The contribution of submerged aquatic vegetation to habitat structure in nearshore marine environments is similarly well recognized. Numerous species use these habitats for cover and rearing during larval and juvenile life-history stages. Submerged aquatic vegetation also provides spawning habitat for Pacific herring.

The degree to which outfalls may modify of the aquatic vegetation community is variable. Buried outfall pipes discharging offshore may have a limited effect on the aquatic vegetation community following recovery from construction impacts. In contrast, exposed outfall pipes may affect vegetation community structure through hydraulic and geomorphic effects imposed on the nearshore environment. Outfall discharges may cause alteration of the aquatic vegetation community through the introduction of toxics or through eutrophication induced by nutrient loading. Alterations of the submerged aquatic vegetation community through reduction in aerial extent or conversion to other habitat types (e.g., conversion of eelgrass habitat to algae and kelp) can reduce the productivity of these habitats for dependent life-history stages. Applying a worst-case scenario perspective, outfalls could result in the long-term alteration of the nearshore aquatic vegetation community through their effects on habitat structure and water quality. This equates to a high risk of take for species dependent on these habitats due to long-term effects on spawning productivity, as well as larval survival, growth, and fitness.

9.4.6 Water Quality Modifications

The effects of outfall construction on water quality modification and related risk of take are similar to that described for dams in Section 9.1.6 (*Water Quality Modifications*), but the construction effects associated with outfalls are of lesser magnitude and duration.

Once operational, most outfalls become vectors for the delivery of pollutants into surface waters. Stormwater and effluent discharges may contain a variety of toxic substances or other pollutants, including PAHs, metals, agricultural chemicals, and nutrients. Alteration of water quality conditions is associated with long-term detrimental effects on the survival, growth, and fitness of aquatic species exposed to the component stressors. Eutrophication caused by nutrient inputs may ultimately lead to decreased DO levels and altered pH conditions, also having potential effects on the survival, growth, and fitness of aquatic receptors. Exposure to these stressors is equated with a high risk of take based on the potential for long-term, chronic exposure.

9.5 Intakes and Diversions

Water diversion and water intake structures include a broad range of designs with purposes ranging from municipal and irrigation water diversions, to power plant and industrial water intakes, to hatchery water supply systems. Structure designs associated with these types of facilities can range from bankline intake systems oriented parallel to the shoreline in any environment type, to dam or weir type diversion structures in river systems oriented perpendicular to streamflow. For the purpose of assessing the risk of take, a worst-case scenario perspective is applied regarding the potential extent of project impacts in riverine and lacustrine/marine environment types. In riverine environments, the worst-case scenario design is a cross-channel type diversion structure similar to a dam or a weir. In marine and lacustrine environments, the worst-case scenario design is a bankline structure similar in magnitude to large tide gates or similar structures.

A significant impact mechanism caused by diversion structures and water intakes that is not addressed in this white paper is the entrainment of slow-swimming or planktonic life-history stages into intake or diversion systems, or the impingement of small fish on diversion or intake screen systems. This impact mechanism is addressed in the Fish Screens white paper (Herrera 2007b).

Species-specific risk of take ratings for diversion structure and water intake development and operation are presented by impact mechanism in Table 9-5. The specific stressors and related risk of take from impact mechanisms caused by this subactivity type are described by environment type in the following subsections.

9.5.1 Construction and Maintenance

Diversion structures and water intakes vary tremendously in size and scale, depending on the purpose of the structure and the environment in which it is placed. As discussed, for the purpose of evaluating risk of take, the impact mechanisms imposed by riverine diversion structures are assumed to be of the same scale as those of a channel-spanning weir structure. Impact mechanisms for the construction and maintenance of intake structures in lacustrine and marine environments are assumed to be of the same scale as those for a bankline structure similar to a large tide gate.

9.5.1.1 Riverine Environments

The construction and maintenance related submechanisms of impact and related risk of take imposed by water diversion and intake structures in riverine environments are similar to those for weirs (see Section 9.2.1 [*Construction and Maintenance*]).

9.5.1.2 Lacustrine and Marine Environments

The construction and maintenance related submechanisms of impact and related risk of take imposed by water diversion and intake structures in riverine environments are similar to those for tide gates and flood gates (see Section 9.6.1 [*Construction and Maintenance*]).

9.5.2 Hydraulic and Geomorphic Modifications

The degree to which diversion or intake structures modify hydraulic and geomorphic conditions in the aquatic environment is a function of the scale of the structure and how it interacts with these physical processes. This type of structure is often configured differently in riverine, marine, and lacustrine environments based on differences in the types of projects typically implemented in these environment types, and to accommodate differences in hydraulic and geomorphic processes. Therefore, the risk of take associated with this impact mechanism is addressed separately for each environment type.

9.5.2.1 Riverine Environments

Hydraulic and geomorphic modification submechanisms of impact and related risk of take imposed by water diversion and intake structures are similar to those for weirs (see Section 9.2.2 [*Hydraulic and Geomorphic Modifications*]).

9.5.2.2 Lacustrine and Marine Environments

Hydraulic and geomorphic modification submechanisms of impact and related risk of take imposed by water diversion and intake structures in lacustrine and marine environments are similar to those for tide gates (see Section 9.6.2 [*Hydraulic and Geomorphic Modifications*]).

9.5.3 Ecosystem Fragmentation

Water diversion and intake systems can cause ecosystem fragmentation through a number of pathways. Dam-like structures in riverine environments can have significant effects on ecological connectivity in the longitudinal, lateral, and vertical dimensions through the effects of the structure on hydraulic and geomorphic conditions, as well as the influence of water withdrawals on stream habitat structures. In lacustrine and marine environments, these effects are less pronounced, but changes in shoreline habitat conditions caused by the alteration of longshore drift, current, and circulation patterns imposed by intake structures, as well as the structure itself, can alter the suitability of nearshore habitats for HCP species that occur in these environments. Fragmentation of migratory corridors is a potential result of these effects.

9.5.3.1 Riverine Environments

Ecosystem fragmentation submechanisms of impact and related risk of take imposed by water diversion and intake structures are expected to be similar to those for dams and weirs (see Section 9.1.3 [*Ecosystem Fragmentation*] under Section 9.1 [*Dams*]).

9.5.3.2 Lacustrine and Marine Environments

Ecosystem fragmentation submechanisms of impact and related risk of take imposed by water diversion and intake structures in lacustrine and marine environments are similar to those for tide gates (see Section 9.6.3 [*Ecosystem Fragmentation*]).

9.5.4 Riparian Vegetation Modifications

Riparian vegetation modification associated with water diversion and intake structures occurs through two primary pathways: riparian impacts occurring during project construction (including loss of vegetation in the structural footprint and access points), and the effects of water withdrawals on riparian habitat in downstream reaches. The latter source of effects is expected to operate only in riverine environments.

9.5.4.1 Riverine Environments

Riparian vegetation modification submechanisms of impact and related risk of take imposed by water diversion and intake structures are expected to be similar to those for weirs (see Section 9.2.4 [*Riparian Vegetation Modifications*]).

9.5.4.2 Lacustrine and Marine Environments

Riparian vegetation modification submechanisms of impact and related risk of take imposed by water diversion and intake structures in lacustrine and marine environments are similar to those for tide gates (see Section 9.6.4 [*Riparian Vegetation Modifications*]).

9.5.5 Aquatic Vegetation Modifications

Aquatic vegetation modification associated with water diversion and intake structures occurs through two primary pathways: vegetation impacts occurring during project construction and maintenance (including loss of vegetation in the structural footprint), and the effects of water withdrawals on aquatic vegetation in the source body. The latter mechanism of effect is expected to operate only in riverine environments.

9.5.5.1 Riverine Environments

Aquatic vegetation modification submechanisms of impact and related risk of take imposed by water diversion and intake structures are expected to be similar to those for weirs (see Section 9.2.5 [*Aquatic Vegetation Modifications]*).

9.5.5.2 Lacustrine and Marine Environments

Aquatic vegetation modification submechanisms of impact and related risk of take imposed by water diversion and intake structures in lacustrine and marine environments are similar to those for tide gates (see Section 9.6.5 [*Aquatic Vegetation Modifications*]).

9.5.6 Water Quality Modifications

Water quality modifications associated with water diversion and intake structures occur through two primary pathways: water quality impacts occurring during project construction, and the effects of water withdrawals on water quality in the source body. The latter mechanism of effects is expected to operate only in riverine environments.

9.5.6.1 Riverine Environments

Water quality modification submechanisms of impact and related risk of take imposed by water diversion and intake structures are expected to be similar to those for dams (see Section 9.1.6 [*Water Quality Modifications*]).

9.5.6.2 Lacustrine and Marine Environments

Water quality modification submechanisms of impact and related risk of take imposed by water diversion and intake structures in lacustrine and marine environments are similar to those for tide gates, with the exception of the introduction of toxic substances (see Section 9.6.6 [*Water Quality Modifications*]). Stressors associated with this impact mechanism would only be expected to occur during project construction, before the structure begins discharging effluent. Therefore, the risk of take resulting from this submechanism would be short term in duration.

9.6 Tide Gates

Tide gates and flood gates are structures designed to facilitate the flow of water out of floodplain, wetland, or estuarine habitats, as well as manage or prevent the reflooding of these lands by tidal fluctuations or flood flows. Tide gates and flood gates range in scale from simple, corrugated metal culverts with metal or fiberglass flap gates buried in dikes, to larger, more complex wood or concrete structures with mechanically controlled gates. They are typically incorporated into dikes and levees to promote the conversion of these habitat types into terrestrial or modified aquatic environment types for human uses. In some cases, tide gates are used to manage habitat conditions within an impounded area to support recreational fish and wildlife populations, but in many cases these structures are intended to facilitate the conversion of estuarine or floodplain wetlands to terrestrial habitats for agricultural or industrial uses. For the purpose of assessing risk of take, a worst-case scenario perspective is taken with regards to the magnitude of construction impacts and the extent of the habitat modifications imposed by these structures.

9.6.1 Construction and Maintenance

Construction of complex tide gate structures may involve in-water equipment use and materials placement, creation of exclusion areas, and potentially pile driving. The impact mechanisms imposed by tide gate construction and maintenance are similar to those described for dams in Section 9.1.1.1 [*Elevated Underwater Noise*] but are of lesser magnitude due the smaller scale of tide gate structures. The specific stressors and related risk of take from impact mechanisms caused by this subactivity type are described in the following subsections. Anticipated submechanisms and related stressors are described below. Species-specific risk of take ratings for tide gates and flood gates are presented by impact mechanism in Table 9-6.

When considering the risk of take ratings presented here and in Table 9-6, it is important to recognize that tide gate construction will in most, if not all, cases take place in environments that are already highly modified by dikes and levees. Degraded channel and bank conditions may not present suitable habitat for HCP species and life-history stages that would otherwise occupy the affected environment. Therefore, while the risk of take ratings are representative of the effects of stressor exposure, the potential for stressor exposure is likely to be more limited than for subactivity types implemented in more pristine environments.

9.6.1.1 Elevated Underwater Noise

The risk of take associated with this submechanism is similar to that described for dams in Section 9.1.1.1 (*Elevated Underwater Noise*). Tide gate construction and maintenance may include pile driving, a source of intense underwater noise with the potential to cause disturbance, injury, or even direct mortality in fish and possibly in invertebrates. Due to the potential for injury and mortality, the risk of take associated with underwater noise is rated as high for species with life-history stages that occur in environments suitable for tide gates. However, the potential for stressor exposure is more limited because tide gate construction would typically be expected to be more limited and to take less time than dam construction.

9.6.1.2 Equipment Operation and Materials Placement

The risk of take associated with this submechanism is similar to that described for dams in Section 9.1.1.2 (*Equipment Operation and Materials Placement*), but operates at a lesser degree of magnitude. In a worst case scenario, tide gate construction and maintenance may involve in-water work, including equipment use and material placement. These activities could result in potential injury or mortality of HCP species occurring in the vicinity that have sessile or non-motile life-history stages. These effects are equated with a high risk of take. Motile species or those with motile life-history stages would experience temporary disturbance and displacement, potentially affecting survival, growth, and productivity. These effects are equated with a moderate risk of take.

9.6.1.3 Bank/Channel/Shoreline Disturbance

Tide gate construction by definition involves bank, channel, and/or shoreline disturbance. Risk of take associated with this submechanism is similar to that described for dams in Section 9.1.1.3 (*Bank/Channel/Shoreline Disturbance)*, but presents a much smaller degree of risk. In a worst case scenario, tide gate construction may require significant disturbance of the bank/shoreline and substrate, degrading habitat conditions in the affected habitat and resulting in the release of suspended sediments. These activities could result in potential injury or mortality of HCP species having sessile or non-motile life-history stages. These effects are equated with a high risk of take. Motile species or those with motile life-history stages would experience temporary disturbance and displacement, potentially affecting survival, growth, and productivity. These effects are equated with a moderate risk of take.

9.6.1.4 Dewatering, Flow Bypass, and Fish Handling

Tide gate construction may require temporary dewatering and/or flow bypass during construction. The risk of take associated with this submechanism is similar to that described for dams in Section 9.1.1.4 (*Dewatering, Flow Bypass, and Fish Handling)*, but tide gates present a smaller magnitude of risk due to the smaller size of the construction footprint. Creation of exclusion areas, fish removal and relocation, and work area dewatering/flow bypass are all activities with the potential to cause injury or mortality to HCP species. Therefore, the effects of this submechanism are equated with a high risk of take.

9.6.2 Hydraulic and Geomorphic Modifications

Tide gates operate in concert with dikes and levees to modify aquatic landscapes for managed human uses. The primary mechanisms through which this occurs are the modification of local hydraulic conditions.

9.6.2.1 Altered Flow Regime (Tidal and Floodwater Exchange)

Alteration of tidal and/or floodwater exchange is the primary submechanism through which tide gates impose their effects on aquatic systems. These structures control or prevent the inflow of tidal fluctuations and floodwaters landward of the structure, while allowing surface flows to drain from landward areas during low-water conditions waterward of the structure. This decreases inundation frequency in what are typically floodplain, wetland, or estuarine marsh habitats, fundamentally altering the characteristics of these landscapes. The stressors imposed by this mechanism are significant and are primarily realized through ecosystem fragmentation (described in Section 9.6.3 [*Ecosystem Fragmentation*]). However, tide gates also concentrate and thereby accelerate the rate at which floodwaters drain from inundated habitats. This change in flow regime may cause the displacement of small or relatively non-motile species adapted to slow-water environments. For example, Olympic mudminnows are adapted to slack water environments in slow-moving streams and wetland ponds. Accelerated flows draining the wetland and stream system caused by the installation of a dike and flood gate system could lead to the displacement of this slow-swimming species, potentially to a riverine environment with unsuitable habitat conditions. In such special cases, mortality would be likely. This submechanism would therefore be equated with a high risk of take, but this stressor would be considered a relatively minor component of the overall impacts of the conversion of floodplain wetland habitat into a managed terrestrial habitat.

9.6.2.2 Altered Channel Geometry

Tide gates and flood gates alter channel geometry through several pathways. First, the structure can force scouring, deposition, and simplification of channel structure by changing inundation frequency and flow velocities in channel networks landward of the structure. Second, by encouraging sedimentation of the channel network over time, distributary channels and ponds gradually fill and become terrestrial habitat (or are converted to managed ditches that are dredged). This will invariably alter the habitat suitability and productivity for HCP species adapted to this type of environment, and these effects will be long term and progressive in nature. This submechanism is equated with a high risk of take, with this stressor considered to be one component of the broader risk of take resulting from the conversion of aquatic habitat into a managed terrestrial environment.

9.6.2.3 Altered Substrate Composition

Tide gates facilitate the drainage of low-lying estuarine and floodplain habitats, as well as encourage the accumulation of sediments and subsidence landward of the structure. This leads to the gradual sedimentation and filling of distributary channel networks, eliminating these habitat types over time. Waterward of the structure, high-velocity flows out of the tide gate can cause localized scour, mobilizing fine sediments and changing the bed composition. These effects would be limited in scale to a relatively small area waterward of the structure, and would occur in what is already a modified channel environment. Therefore, the additive risk of take associated with this submechanism is considered to be moderate for HCP species with life-history stages that occur in the affected environment.

9.6.3 Ecosystem Fragmentation

Ecosystem fragmentation is the primary mechanism through which tide gates and flood gates operate in concert with dikes and levees to convert aquatic habitats to terrestrial uses. Applying a worst-case scenario perspective, the essential purpose of these structures is to facilitate the flow of water out of floodplain, wetland, or estuarine habitats, while preventing these lands from being reflooded by tidal exchange or flood waters. The alteration and conversion of habitats to conditions that are poorly suited for HCP species are the ultimate results within the zone of effect of the structure, and are long-term in duration. The essentially permanent modification of high-value habitats to unsuitable conditions equates to a high risk of take for those species dependent on these habitats during some portion of their life history.

9.6.4 Riparian Vegetation Modifications

Tide gate construction may require the permanent alteration of riparian vegetation within the footprint of the structure, as well as additional temporary modification of the surrounding habitat during construction. The risk of take associated with this submechanism is similar to that described for *Outfalls* in Section 9.4.4 (*Riparian Vegetation Modifications*), due to the similar size of the construction footprint. However, it is important to consider that these structures are typically developed in environments where riparian conditions have already been extensively modified for dike and levee development; therefore, the actual risk of take associated with this impact mechanism may be insignificant in comparison to that imposed by the hydromodification.

9.6.5 Aquatic Vegetation Modifications

The effects of tide gates and flood gates on aquatic vegetation and the resulting risk of take are similar to those described for dikes and levees. These effects are compounded by water quality related effects exacerbated by the exposure of anaerobic sediments in floodplain and estuarine environments (see Section 9.6.6 [*Water Quality Modifications*]). This submechanism is equated with a high risk of take for those HCP species dependent on floodplain, wetland, and estuarine marsh habitats during some portion of their life history, particularly species such as Newcomb's littorine snail that are obligate occupants of emergent saltmarsh vegetation.

9.6.6 Water Quality Modifications

Tide gate and flood gate installation imposes a number of water quality related submechanisms beginning with construction and proceeding as the aquatic ecosystem landward of the structure adjusts to the new conditions imposed on its physical processes. Alteration of flow conditions and inundation frequency lowers surface and groundwater levels, exposing large amounts of organic material sequestered in an anaerobic environment to aerobic decomposition, thereby unleashing a cascade of changes in surface water chemistry. This can lead to a risk of take through a variety of submechanisms of impact. While the risk of take associated with each of these submechanisms is discussed separately here, these effects are interrelated and the stressors they produce act synergistically on exposed receptors. Therefore, the risk of take resulting from each of these individual submechanisms should be considered collectively.

9.6.6.1 Altered Temperature Regime

Tide gates alter the ambient water temperature regime in aquatic environments landward of the structure by limiting the exchange and flushing effects of tidal inundation and floodwaters. These effects occur predominantly in tidally influenced areas where the flushing effects of tidal exchange normally occurs on a daily basis. In such circumstances, aquatic habitats landward of the structure would be expected to experience elevated water temperatures, particularly during summer months. Organisms exposed to chronic elevations in water temperatures beyond tolerance thresholds would be expected to experience reduced survival, growth, and fitness. Due to the essentially permanent nature of these effects, this submechanism is equated with a high risk of take.

9.6.6.2 Altered Salinity

By virtue of design and intended function, tide gates alter the salinity of surface waters upstream of the structure. Tide gates prevent the tidal inflow of marine water, resulting in conversion to freshwater habitat over time as freshwater inflow changes the characteristics of the system. This wholesale conversion from estuarine or marine to freshwater habitat represents a fundamental alteration in habitat suitability for species adapted to the original habitat conditions. Because these effects will persist for the life of the structure, they are associated with a high risk of take for HCP species that utilize environments suitable for tide gate development.

9.6.6.3 Altered Dissolved Oxygen

Alteration of flow regime and inundation frequency in saltmarsh and wetland environments has been demonstrated to cause depleted oxygen conditions as organic matter in anoxic soils becomes exposed and available for aerobic decomposition. These combined effects have been demonstrated in saltmarsh ecosystems regulated by tide gates to deplete DO concentrations below levels sufficient to cause direct mortality of fish. Even in the absence of mortality, stress from DO depletion in combination with increased water temperatures and poor habitat suitability may lead to decreased survival, growth, and fitness of HCP species occurring within the modified habitat. Freshwater wetland environments would be expected to experience similar effects, where the operative physical, biological, and chemical processes are similar. Due to their long-term and progressive nature, these effects are equated with a high risk of take for species occurring in the affected environment.

9.6.6.4 Altered Suspended Solids and Turbidity

Tide gate and flood gate construction creates the likelihood of increased suspended sediment levels in nearshore environments from channel and bank or shoreline disturbance during equipment use and materials placement. The risk of take from this construction-related submechanism is similar to that described for dams in Section 9.1.6.3 (*Altered Suspended Sediments and Turbidity*).

9.6.6.5 Altered pH

Some tide gate and flood gate structures are built using concrete, a material capable of causing acute changes in surface water pH if appropriate best management practices are not employed during construction. The risk of take from this construction-related submechanism is similar to that described for dams in Section 9.1.6.7 [*Introduction of Toxic Substances*].

Once a tide gate or flood gate is in place, the alteration in inundation frequency describe above can lead to the exposure of anaerobic sediments to open air. Oxidation of sulfides released from anaerobic sediments can in turn rapidly reduce the pH of surface waters. This effect is well documented in the literature in natural systems, and may be compounded in environments that are undergoing a conversion to terrestrial habitat imposed by a dike/tide gate system. Rapid reductions in pH are capable of causing physiological stress, injury, and mortality in many fish and invertebrate species. Therefore, this submechanism is equated with a high risk of take.

9.6.6.6 Altered Nutrient Loading

Tide gate and flood gate construction presents the opportunity for the introduction of toxic substances from accidental spills during project construction. The risk of take from this construction-related submechanism is similar to that described for dams in Section 9.1.6.7 (*Introduction of Toxic Substances*).

Once a tide gate or flood gate is in place, the processes enabled when anaerobic sediments are exposed to oxidation (see Section 9.6.6.3 [*Altered Dissolved Oxygen*] and Section 9.6.6.5 [*Altered pH*]) can cause the release of a number of potentially toxic substances into the aquatic environment. Decreased surface water pH and altered redox conditions in exposed soils can cause rapid leaching of toxic metals, including aluminum, cadmium, copper, and silver into the water column. Decreased pH can, in some cases, produce rapidly precipitating iron flocs capable of smothering wildlife and vegetation. Water quality modifications initially occur landward of the structure but can extend beyond the dike into the nearshore environment as the altered surface water drains during low tide or low streamflow conditions. These kinds of effects are well documented in natural systems and may be compounded in environments that are undergoing a relatively rapid conversion to terrestrial habitat imposed by a dike/tide gate system.

Exposure to dissolved metals and floc precipitates can impose physiological stress, injury, and mortality on HCP species exposed to these stressors. These stressors may also weaken or kill aquatic vegetation, altering habitat structure and suitability for organisms dependent on these habitat types. Due to the potential for direct mortality and the intermediate to long-term nature of these effects, this submechanism is equated with a high risk of take.

Table 9-1. Species- and habitat-specific risk of take for mechanisms of impacts associated with dams.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Coho salmon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Chum salmon	H	N	I	H	N	I	H	N	I	H	N	I	M	N	I	H	N	I
Pink salmon	H	N	I	H	N	I	H	N	I	H	N	I	M	N	I	H	N	I
Sockeye salmon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Steelhead	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Coastal cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Redband trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Westslope cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Bull trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Dolly Varden	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Pygmy whitefish	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Olympic mudminnow	N	N	N	H	N	H	H	N	H	N	N	N	N	N	N	N	N	N
Lake chub	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Leopard dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Margined sculpin	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Umatilla dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Pacific lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
River lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Western brook lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
White sturgeon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H

Table 9-1 (continued). **Species- and habitat-specific risk of take for mechanisms of impacts associated with dams.**

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Longfin smelt	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Eulachon	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Table 9-1 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with dams.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb’s littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H
Western ridged mussel	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-2. Species- and habitat-specific risk of take for mechanisms of impacts associated with weirs.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	N	H	H	N	M	H	N	H	I	I	I	M	N	M	H	N	H
Coho salmon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Chum salmon	H	N	I	H	N	I	H	N	I	H	N	I	M	N	I	H	N	I
Pink salmon	H	N	I	H	N	I	H	N	I	H	N	I	M	N	I	H	N	I
Sockeye salmon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Steelhead	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Coastal cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Redband trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Westslope cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Bull trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Dolly Varden	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Pygmy whitefish	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Olympic mudminnow	N	N	N	H	N	H	H	N	H	N	N	N	N	N	N	N	N	N
Lake chub	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Leopard dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Margined sculpin	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Umatilla dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Pacific lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
River lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Western brook lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
White sturgeon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H

Table 9-2 (continued). **Species- and habitat-specific risk of take for mechanisms of impacts associated with weirs.**

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Longfin smelt	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Eulachon	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Table 9-2 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with weirs.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H
Western ridged mussel	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-3. Species- and habitat-specific risk of take for mechanisms of impacts associated with dikes and levees.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Chinook salmon	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Coho salmon	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Chum salmon	H	H	I	H	H	I	H	H	I	H	H	I	M	M	I	H	H	I
Pink salmon	H	H	I	H	H	I	H	H	I	H	H	I	M	M	I	H	H	I
Sockeye salmon	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Steelhead	H	M	H	H	?	M	H	?	H	H	?	M	M	?	M	H	M	H
Coastal cutthroat trout	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Redband trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Westslope cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Bull trout	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Dolly Varden	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Pygmy whitefish	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Olympic mudminnow	N	N	N	H	N	H	H	N	H	N	N	N	N	N	N	N	N	N
Lake chub	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Leopard dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Margined sculpin	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Umatilla dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Pacific lamprey	H	I	H	H	I	M	H	I	H	H	I	M	M	I	M	H	I	H
River lamprey	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Western brook lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N
White sturgeon	H	?	H	H	?	M	H	?	H	H	?	M	M	?	M	H	?	H

Table 9-3 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with dikes and levees.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Longfin smelt	H	H	N	H	H	N	H	N	N	M	I	N	I	?	?	H	H	N
Eulachon	H	H	N	H	H	N	H	N	N	M	I	N	I	?	N	H	H	N
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Surf smelt	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Pacific herring	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Lingcod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Pacific cod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Pacific hake	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Walleye pollock	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Black rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Brown rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Canary rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
China rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Copper rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Widow rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Olympia oyster	N	H	N	N	H	N	N	N	N	N	I	N	N	I	N	N	H	N

Table 9-3 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with dikes and levees.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb's littorine snail	N	H	N	N	H	N	N	H	N	N	H	N	N	N	N	N	L	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H
Western ridged mussel	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-4. Species- and habitat-specific risk of take for mechanisms of impacts associated with outfalls.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Coho salmon	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Chum salmon	H	H	I	M	H	I	H	H	I	I	I	I	I	H	I	H	H	I
Pink salmon	H	H	I	M	H	I	H	H	I	I	I	I	I	H	I	H	H	I
Sockeye salmon	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Steelhead	H	M	H	H	?	H	H	?	H	H	?	I	I	?	I	H	?	H
Coastal cutthroat trout	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Redband trout	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Westslope cutthroat trout	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Bull trout	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Dolly Varden	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Pygmy whitefish	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Olympic mudminnow	H	N	H	M	N	H	H	N	H	I	N	I	N	N	N	H	N	H
Lake chub	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Leopard dace	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Margined sculpin	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Umatilla dace	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Pacific lamprey	H	I	H	M	I	H	H	I	H	I	I	I	I	I	I	H	I	H
River lamprey	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Western brook lamprey	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N
White sturgeon	H	?	H	H	?	M	H	?	H	H	?	I	M	?	M	H	?	H

Table 9-4 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with outfalls.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Longfin smelt	H	H	H	M	H	H	H	H	H	I	I	I	I	?	I	H	H	H
Eulachon	H	H	N	M	H	N	H	H	N	I	I	N	I	?	N	H	H	N
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Surf smelt	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Pacific herring	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Lingcod	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Pacific cod	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Pacific hake	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Walleye pollock	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Black rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Brown rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Canary rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
China rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Copper rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Widow rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Olympia oyster	N	H	N	N	H	N	N	N	N	N	I	N	N	I	N	N	H	N

Table 9-4 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with outfalls.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	I	N	N	H	N
Newcomb’s littorine snail	N	H	N	N	H	N	N	N	N	N	H	N	N	N	N	N	L	N
Giant Columbia River limpet	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Great Columbia River spire snail	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	H	N	N
California floater (mussel)	H	N	N	M	N	N	H	N	M	H	N	N	M	N	N	H	N	N
Western ridged mussel	H	N	N	M	N	N	H	N	M	H	N	N	M	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-5. Species- and habitat-specific risk of take for mechanisms of impacts associated with diversion structures and water intakes.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Chinook salmon	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Coho salmon	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Chum salmon	H	H	I	H	H	I	H	H	I	H	H	I	M	M	I	H	H	I
Pink salmon	H	H	I	H	H	I	H	H	I	H	H	I	M	M	I	H	H	I
Sockeye salmon	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Steelhead	H	M	H	H	?	H	H	?	H	H	?	I	M	?	M	H	?	H
Coastal cutthroat trout	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Redband trout	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Westslope cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Bull trout	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Dolly Varden	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Pygmy whitefish	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Olympic mudminnow	N	N	N	H	N	H	H	N	H	N	N	N	N	N	N	N	N	N
Lake chub	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Leopard dace	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Margined sculpin	H	N	N	H	N	N	H	N	N	H	N	I	M	N	N	H	N	N
Mountain sucker	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Umatilla dace	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Pacific lamprey	H	I	H	H	I	M	H	I	H	H	I	I	M	I	M	H	I	H
River lamprey	H	H	H	H	H	M	H	H	H	H	I	I	M	H	M	H	H	H
Western brook lamprey	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N
White sturgeon	H	?	H	H	?	M	H	?	H	H	?	I	M	?	M	H	?	H

Table 9-5 (continued). **Species- and habitat-specific risk of take for mechanisms of impacts associated with diversion structures and water intakes.**

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Longfin smelt	H	H	N	H	H	N	H	N	N	M	I	N	I	?	N	H	I	N
Eulachon	H	H	N	H	H	N	H	N	N	H	I	N	I	?	N	H	I	N
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Surf smelt	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Pacific herring	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Lingcod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Pacific cod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Pacific hake	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Walleye pollock	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Black rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Brown rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Canary rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
China rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Copper rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Widow rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Olympia oyster	N	H	N	N	H	N	N	N	N	N	I	N	N	I	N	N	H	N

Table 9-5 (continued). **Species- and habitat-specific risk of take for mechanisms of impacts associated with diversion structures and water intakes.**

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb's littorine snail	N	H	N	N	H	N	N	H	N	N	H	N	N	N	N	N	L	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H
Western ridged mussel	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-6. Species- and habitat-specific risk of take for mechanisms of impacts associated with tide gates.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Chinook salmon	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Coho salmon	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Chum salmon	H	H	I	H	H	I	H	H	H	I	I	I	M	H	I	H	H	H
Pink salmon	H	H	I	H	H	I	H	H	H	I	I	I	M	H	I	H	H	H
Sockeye salmon	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Steelhead	H	H	H	H	?	H	H	?	H	I	?	I	M	?	M	H	M	H
Coastal cutthroat trout	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Redband trout	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Westslope cutthroat trout	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Bull trout	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Dolly Varden	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Pygmy whitefish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympic mudminnow	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Lake chub	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Leopard dace	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Mountain sucker	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Umatilla dace	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Pacific lamprey	H	I	H	H	I	H	H	I	H	I	I	I	M	I	M	H	M	H
River lamprey	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Western brook lamprey	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N
White sturgeon	H	?	H	H	?	H	H	?	H	H	?	I	M	?	M	H	?	H

Table 9-6 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with tide gates.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Longfin smelt	H	H	H	H	H	H	H	H	H	I	I	I	I	?	?	H	H	H
Eulachon	H	H	N	M	H	N	H	H	N	I	I	N	I	?	N	H	H	N
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Surf smelt	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Pacific herring	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Lingcod	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Pacific cod	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Pacific hake	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Walleye pollock	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Black rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Brown rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Canary rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
China rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Copper rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Widow rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Olympia oyster	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N

Table 9-6 (continued). Species- and habitat-specific risk of take for mechanisms of impacts associated with tide gates.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	H	N
Newcomb's littorine snail	N	H	N	N	H	N	N	H	N	N	H	N	N	N	N	N	L	N
Giant Columbia River limpet	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
California floater (mussel)	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Western ridged mussel	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

10.0 Data Gaps

This section identifies data gaps and the information needed to fill those gaps for the construction and maintenance of flow control structures. Several data gaps extend across all six subactivities covered in this white paper and are addressed first. Additional subactivity-specific data gaps are then discussed. Finally, it should be noted that there is a general lack of cumulative impact studies for flow control structures and therefore is considered a data gap. The one exception is dams, where several studies have documented the cumulative effects on fish migrations when a series of dams are present. These are addressed in Section 8 (*Cumulative Effects*).

10.1 Common Data Gaps

Several data gaps have been identified that are universal for all six subactivity types addressed in this white paper: impacts from construction and maintenance activities, impacts from altered riparian vegetation in marine environments, habitat effects on fish species, and specific information on impacts for several HCP species, in particular information on invertebrates.

10.1.1 Construction and Maintenance Activities

In general, there is a lack of information or direct studies associated with construction-related impacts on HCP species for all subactivity types. It is expected that during the construction of flow control structures there will be: increased noise from non-pile driving activities; dewatering and fish handling; and increased erosion and sediment transport (from dredging). While the potential for these impacts to occur is described in Section 7 (*Direct and Indirect Impacts*), no studies have examined these changes during the construction of flow control structures. These data gaps are described in more detail in the following sections.

10.1.1.1 Elevated Underwater Noise

Exposure to pile driving noise is likely the primary source of underwater noise that is known to cause mortality and injury to some HCP species; however, the effects of underwater noise on mollusks are currently a data gap. Additional research is needed on this topic to evaluate noise impacts generated by various equipment types on a diversity of species, including shellfish. Data gaps on the hearing capacities of HCP species and the effects of increased underwater noise on hearing as well as the heart, kidneys, and other highly vascular tissue due to flow control structure construction remain. Although studies have identified elevated hearing thresholds in response to engine and other white noises for cyprinid fishes (which are hearing specialists), data are needed on hearing (as well as the heart, kidneys, and other highly vascular tissue) thresholds and effects on HCP species. In addition, data gaps exist on the temporary, chronic, and cumulative affects of underwater noise induced by construction of flow control structures in marine, riverine, and lacustrine environments.

10.1.1.2 Dewatering, Flow Bypass, and Fish Handling

Few studies have compared the susceptibility of various fish and macroinvertebrate species to different types of handling techniques. More information comparing the susceptibility to injuries associated with these types of techniques is needed to identify potential take for these species. Training and minimum qualifications for personnel performing fish capture and handling (particularly electrofishing) are also needed to define standard protocols that would minimize take. Most of the studies on the effects of fish handling have been performed on electrofishing. Electrofishing effects have been conducted on adult fish greater than 12 inches in length (Dalbey et al. 1996). The relatively few studies that have been conducted on juvenile salmonids indicate that spinal injury rates are substantially lower than they are for large fish. Only a few recent studies have examined the long-term effects of electrofishing on salmonid survival and growth (e.g., Ainslie et al. 1998, Dalbey et al. 1996). Little research has been conducted on the effects of dewatering and fish capture and handling on nonsalmonid HCP species. More directed research is necessary to understand the risk of take resulting from this submechanism for these species.

10.1.1.3 Construction/Maintenance Dredging

There are numerous studies of impacts on aquatic species from dredging activities (Cooper et al. 2007; Erftemeijer and Lewis 2006; Newell et al. 2004). However, these impacts have been shown to be site- and species-specific (Byrnes et al. 2004), with “opportunistic” species (e.g., mollusks) being much less affected than those that have long life histories (e.g., rockfish) (Newell et al. 2004). Considering the diversity of environments present in Washington, a number of data gaps exist with respect to specific HCP species, particularly with the effects on rockfish from adjacent dredging operations. While dredging is already prohibited in rockfish nursery areas by WAC 220-110-320, adjacent areas potentially exposed to heightened turbidity are not covered by this legislation. Turbidity thresholds that have been used successfully in existing monitoring programs to protect aquatic species are unknown and are considered a data gap (Thorkilsen and Dynesen 2001).

Although the physics of turbidity generation can be calculated, adequate data do not exist to quantify the biological response in terms of threshold sediment dosages and exposure durations that can be tolerated by each of the HCP species. Numerical modeling simulations of dredging-related suspended sediment plume dynamics need to be correlated with field and laboratory studies to further identify information needs on each of the HCP species. In marine environments, existing data indicate that responses to suspended sediments are highly species-specific, with some species having lethal effects at several hundred ppm in 24 hours and others having no effect at concentrations above 10 ppt for 7 days. Studies on East Coast species have identified lethal suspended concentration levels, and Newcombe and Jensen (1996) developed a predictive model for defining lethal and sublethal fish injury threshold levels for suspended sediment concentrations. However, threshold studies (single-event as well as cumulatively) are lacking for the temporary impacts of suspended sediment levels specific to dredging in Pacific Northwest marine, lacustrine, and riverine environments (Nightingale and Simenstad 2001a).

The following information needs are also considered data gaps:

- Comprehensive data on the spatial and temporal distribution of spawning, rearing, and migration behaviors of HCP species to determine and assign dredging work windows on a site-specific basis have not been compiled.
- Cumulative thresholds associated with dredge-induced changes in salinity intrusion and other critical physicochemical processes in marine environments have not been identified.
- Recovery capability for HCP species that may be at risk of impacts from temporary exposure, chronic exposure, and cumulative thresholds associated with dredging in marine, lacustrine, and riverine environments in early life-history as well as adult stages are not fully understood for many HCP species.
- Recolonization capacities, after temporary, chronic, or cumulative thresholds are reached, of HCP species and the species endemic to those habitats (in marine, lacustrine, and riverine environments) that are important to their growth and survival are not yet understood.
- Temporary, chronic, and cumulative effects associated with nighttime lighting from dredge equipment (during construction as well as during operations following construction) have not been comprehensively investigated. The role of lighting in attracting predator species to affected sites is not fully understood.
- The magnitude and duration of noise associated with dredging operations have not been evaluated. Additional research on fish responses to noise is needed. This information is needed to evaluate potential noise impacts on HCP species.

Fish behavior responses to dredging-related turbidity plumes of different extents are not yet understood.

10.1.2 Marine Riparian Vegetation Modifications

Although the functions of freshwater riparian vegetation have been identified for riverine systems, exploring and defining the functions of marine riparian vegetation are ongoing. There is reason to believe that marine riparian vegetation provides similar functions to riparian vegetation adjacent to freshwater habitats; however, the extent and nature of those functions are not fully understood (Desbonnet et al. 1995; NRC 2001; Brennan and Culverwell 2004). The following information needs are outstanding: (1) understanding the specific nature and function of riparian habitat elements along marine shorelines; (2) the dependence of HCP species on riparian marine and freshwater habitat functions; (3) the effects on HCP species from flow control modifications to those habitats; (4) the dependence of HCP species on marine allochthonous inputs; and (5) the cumulative and synergistic effects of riparian and shoreline removal.

Furthermore, no research has been conducted to study submarine and intertidal groundwater in Puget Sound. It is clear from work elsewhere that such flows are crucial in sustaining nearshore ecosystems (Gallardo and Marui 2006); however, their role on the nearshore environment throughout Puget Sound is virtually unknown (Finlayson 2006).

10.1.3 Habitat Effects on Species

One of the biggest gaps in the literature is information that directly relates habitat changes to fish productivity (Bolton and Shellberg 2001). This information is difficult to collect because multiple factors may simultaneously influence the overall productivity and survival of fish species. However, this type of information is crucial to understand to minimize impacts on HCP species from flow control structures.

10.1.4 HCP Species-Specific Information

Besides the extensive research on salmonids, there is a general lack of information regarding the effects that flow control structures may have on other HCP fish species. An exception is that several studies have been conducted examining sturgeon and dams. In addition, during a detailed literature review for each subactivity type, little information on direct impacts on invertebrates was found. As with construction, impacts can be implied with a good deal of certainty (e.g., increased sediments will adversely affect invertebrates); however, more research is warranted, especially for Pacific Northwest species. For example, there are no studies on the harm threshold(s) of suspended sediments on species in the Pacific Northwest (see Section 10.1.1.3 for more information).

10.1.5 Lost-Opportunity Impacts

Although it is recognized that lost-opportunity impacts must be mitigated to achieve no loss of habitat (WDFW 2003), currently there are no tools for universal and consistent application of the concept. Tools are needed to assess the lost opportunities associated with flow control structures to ensure that appropriate mitigation is provided.

10.2 Dams

Impacts from dams have been well documented and has been a topic of research for decades. In general, there are no major data gaps that exist. However, little research on the hyporheic zone has been conducted in highly altered and degraded fluvial systems (Bolton and Shellberg 2001).

As the understanding of these processes increases, studies will likely begin to focus more on the effects of land use and other human activities on surface–groundwater interactions.

Finally, the effects of dam removal on aquatic species, their habitats, and ecological processes represent a data gap. Although there have been several studies on the ecological impacts of dam removal (Bednarek 2001), there is a general lack of post-removal data to document these changes. More specifically, dam removal data to date have focused on smaller dams, so the actual impacts from a large dam removal are often inferred. The future removal of two dams on the Elwha River (Washington) represents an opportunity to study the impacts of a large-scale dam removal.

10.3 Weirs

There are no significant data gaps associated with the presence of weirs. However, there is a lack on information on the impacts from weirs on specific HCP species.

10.4 Dikes and Levees

As with dams, little research regarding the hyporheic zone has been conducted in systems supporting dikes and levees. As the understanding of these processes increases, studies will likely focus on the effects of land use and other human activities on surface–groundwater interactions. In addition, while a number of studies document changes to habitat after construction of dikes and levees on the landward side of the structure, more information is needed with respect to in-channel changes.

10.5 Outfalls

Limited information is available on the hydraulic and geomorphic modifications of outfalls and intakes and their direct impact on fish and invertebrates. In addition, the effects of outfalls on riparian and aquatic vegetation and ecosystem fragmentation are scarce. Because field studies on water quality impacts typically capture the effects of these subactivities, it is difficult to identify a specific data gap. In general, most studies of outfalls are related to water quality modifications, and these impacts are well documented.

10.6 Intakes and Diversions

Limited information is available on the hydraulic and geomorphic modifications of intakes and diversions and their direct impact on fish and invertebrates. In addition, the effects of intakes on riparian and aquatic vegetation, hyporheic flows, and ecosystem fragmentation are scarce.

Therefore, general research on these mechanisms of impact from intakes and diversion structures is warranted.

10.7 Tide Gates

In a review of tide gate operations in the Pacific Northwest, Giannico and Souder (2005) failed to find studies that examined the effect of tide gates on juvenile fishes, reporting that this represents a large data gap in our understanding of how these structures influence fish populations. Furthermore, specific information is lacking on migration patterns of species that use habitats where tide gates occur (Giannico and Souder 2005). If detailed information on the behavior and movements of HCP species were better understood, then tide gates could be better designed to allow for increased fish passage.

In addition, there is a potential for a loss of LWD as the result of a tide gate. However, there is little information about how tide gates alter LWD transport and recruitment, which represents a potential data gap. In marine ecosystems in general, the influence of LWD on primary productivity is somewhat unclear.

11.0 Habitat Protection, Conservation, Mitigation, and Management Strategies

The Endangered Species Act requires that impacts on listed species or designated critical habitat be avoided or, if unavoidable, minimized to the maximum extent practicable.

11.1 Common Mitigation Strategies

Several common habitat protection, conservation, and mitigation strategies apply to each flow control structure and include minimization of impacts from: construction and maintenance activities, riparian vegetation modifications, and aquatic vegetation modification.

11.1.1 Construction and Maintenance Activities

In a recent document on procedures to minimize nonpoint source pollution from hydromodification projects, the USEPA (2007) proposed measures to minimize construction problems from sediment increases and chemical pollution. The management practices are specific to the location of the project, the local climate, and source of potential pollution.

Erosion and sediment control procedures are used to prevent sediment from entering surface waters during the construction or maintenance of flow control structures. Proper erosion and sediment control practices should be used to protect surface water quality because of the high potential for the loss of sediment directly to surface waters during these types of projects. Erosion control can be maximized by minimizing the area and time of land disturbance and by stabilizing disturbed soils to prevent erosion in a timely matter. USEPA (2007) has suggested using sediment and erosion control practices borrowed from other applications, such as urban development and construction activities. Potential erosion control activities include application of the following methods and practices:

- Bank shaping and planting
- Bulkheads and seawalls
- Check dams
- Coconut fiber roll
- Erosion control blankets
- Locate potential land disturbing activities away from critical areas
- Mulching
- Preserve on-site vegetation
- Retaining walls
- Revegetation
- Riparian improvements

- Sediment fences
- Sodding
- Vegetated filter strips
- Wind erosions controls.

Minimization of runoff will reduce potential impacts on water quality during construction activities. Practices for controlling chemicals and pollutants include the following (USEPA 2007):

- Check dams
- Constructing runoff intercepts
- Equipment runoff control
- Fuel and maintenance staging areas
- Locate potential land-disturbing activities away from critical areas
- Pesticide and fertilizer management
- Pollutant runoff control
- Preserve on-site vegetation
- Sediment traps
- Vegetated filter strips.

In the construction of new flow control structures, avoidance or minimization of impacts can be accomplished through site selection and facility design. For construction and maintenance activities, management strategies can be implemented to minimize underwater noise, dewatering and fish handling, and construction/maintenance dredging impacts. The following strategies can be used to avoid and, if avoidance is not possible, to minimize and mitigate negative impacts on habitats and HCP species associated with construction and maintenance activities.

11.1.1.1 Pile Driving

- Maintain the integrity of the air bubble curtain; no boat traffic or other structure or equipment should be allowed to penetrate the air curtain during pile driving activities.

- To avoid attracting fishes with lights during nighttime pile driving operations, pile driving should be limited to daylight hours to the extent practicable.

- Pile caps (wood blocks), if feasible and safe, should be used to reduce the sound of pile driving below injury level (Laughlin 2006).

- Vibratory hammers; the low rise in sound over a longer period of time (see Section 7.1.1.1 [*Elevated Underwater Noise*], Figure 7-1) is less stressful to aquatic animals, and the sound is typically 10 to 20 dB lower than impact hammer pile driving (WSDOT 2006).

- For projects with pile sizes less than 24 inches in diameter, use the smallest piling size practicable as they result in lower sound pressure levels when driven.
- Consider using wood or concrete piles where practicable, as these also induce lower sound pressure levels.

11.1.1.2 Elevated Underwater Noise

To protect HCP species from the impacts of increased noise, use noise reduction devises such as:

- Air bubble curtains to create a bubble screen that can reduce peak underwater sound pressure levels by at least 15 dB_{peak} (Reyff et al. 2003; Vagle 2003).
- Maintain the integrity of the air bubble curtain; no boat traffic or other structure or equipment should be allowed to penetrate the air curtain.
- In marine environments, installation of a geotube should occur during low tide to minimize the potential for entrapment and stranding of fish within the enclosed area.
- Use fabric barriers and/or cofferdams to create an additional interface to buffer sound transmission into the underwater environment (WSDOT 2006).

11.1.1.3 Dewatering, Flow Bypass, and Fish Handling

For activities that require dewatering, impacts can be minimized by performing work during low-flow or dry conditions and by pumping sediment-laden water from the work area to an infiltration treatment site. Disturbed areas within the channel should be stabilized with a layer of sediment corresponding to the ambient bed to prevent an influx of fine sediment once water is reintroduced to the site. Science-based protocols for fish removal and exclusion activities should be adopted to track and report the number and species of fish captured, injured, or killed. Projects should also require slow dewatering and passive fish removal from the dewatered area before initiating active fish-removal protocols. During passive fish removal, seining is recommended before resorting to electrofishing, which carries a greater risk of mortality (NMFS 2006).

Further minimize channel dewatering impacts on HCP species by taking the following precautions:

- Perform work during low-flow or dry conditions, and/or during dry weather.
- Pump sediment-laden water (from the work area that has been isolated from surrounding water) to an infiltration treatment site.
- Dispose of debris or sediment outside of the floodplain.
- Stabilize disturbed areas at the work site with sediment corresponding to the ambient bed to prevent an influx of fine sediment once water is reintroduced to the site.
- Adopt science-based protocols for fish removal and exclusion activities, including tracking and reporting of number and species of fish captured, fish injured, and mortality.
- Define and require qualifications for personnel performing fish capture and handling; maintain a list of qualified personnel.
- Require slow dewatering and passive fish removal from the dewatered area before initiating active fish-removal protocols.

11.1.1.3.1 Electrofishing Guidelines

- Require the use of NMFS electrofishing guidelines.
- Use lowest power output for effective electrofishing.
- Use least damaging direct current (not alternating current).
- Watch for burns or brands or muscle spasms as these indicate harm to the fish.
- Use spherical electrodes appropriate to the water conductivity and the desired size and intensity of the field (Snyder 2003).
- Minimize fish exposure to handling by netting rapidly.
- Frequently change holding water to ensure adequate dissolved oxygen levels and avoid excessive temperature rises.
- Avoid crowding of fish in holding areas.

11.1.1.4 Construction /Maintenance Dredging

General recommendations to avoid and minimize the impacts of dredging are provided in the 2001 Dredging: Marine Issues white paper (Nightingale and Simenstad 2001a) and include: (1) more extensive use of multiseason pre- and postdredge project biological surveys to assess animal community impacts; (2) incorporation of cumulative effects analysis into all dredging project plans; (3) increased use of landscape-scale planning concepts to plan for beneficial use projects most suitable to the area's landscape ecology and biotic community and food web relationships; (4) further identification of turbidity and noise thresholds to assess fish injury risks; and (5) further analysis and synthesis of the state of knowledge on what is known about the spatial and temporal distribution of fish and shellfish spawning, migration behavior, and juvenile rearing to evaluate environmental windows for dredging on a site-specific basis.

The following recommendations are intended to reduce the effects of dredging on HCP species:

- For new marine, riverine, and lacustrine projects and significant expansions beyond general maintenance dredging, thoroughly assess the large-scale, cumulative impacts of the resulting changes in bathymetry, habitat loss, and change to estuarine/nearshore marine ecosystem dynamics (e.g., salinity intrusion).

- Require hopper dredges, scows, and barges, trucks or any other equipment used to transport dredged materials to the disposal or transfer sites to completely contain the dredged material.

- For long-term projects where continuous dredging and onloading to barges occur, require periodic movement of the barge to reduce unnecessary shading (for more information regarding shading impacts see the white paper on Marine Overwater Structures [Jones & Stokes 2006]).

- Modify in-water work windows to take into consideration what is known about site-specific spatial and temporal distribution of fish and shellfish eggs, larvae, and juveniles.

- Evaluate the application of in-water work windows on a site-specific basis based on the location and features of the site, such as sediment composition, plant and animal assemblages, and timing of seasonal and migration patterns.

- Use presampling bathymetric surveys, records from previous dredging events, and best professional judgment to estimate the volume of sediments likely to be dredged; base sampling and testing requirements on this estimated volume.

- Avoid projects and expansions that convert intertidal to subtidal habitat. If such conversion is unavoidable, employ comprehensive, large-scale risk assessment to identify the cumulative effects of site-specific changes to ecosystem dynamics.

- Select dredging equipment types according to project-specific conditions, such as sediment characteristics.

- Base turbidity threshold testing for dredging operations on background site turbidity.

- In areas where dredging is proximal to sensitive habitats (or in projects where sediments both suitable and unsuitable for unconfined open water disposal will be dredged adjacent to each other), use the "Silent Inspector" (a computerized electronic sensor system) to monitor dredging operations. This tool can assist in operational documentation and regulatory compliance by providing record accessibility and clarity. It also offers advantages for planning, estimating, and managing dredging activities.

- Increase the use of multiseason, preproject surveys of benthos to compare with postproject surveys to understand dredging impacts.

- Where applicable and involving uncontaminated sediments, consider beneficial use of dredged materials that can contribute to habitat restoration, rehabilitation, and enhancement, particularly for projects that incorporate a landscape ecology approach.

- Avoid beneficial use projects that impose unnatural habitats and features on estuarine, marine, and riverine landscapes.

- Use hydrodynamic models to predict system-wide changes in salinity, turbidity, and other physicochemical regimes for project assessment planning that avoids or minimizes impacts on aquatic habitat.

11.1.2 Riparian Vegetation Modifications

Avoid and minimize any impacts on riparian, aquatic, and shoreline vegetation by protecting the vegetation. If it is not possible to leave vegetation, prepare revegetation plans to restore the riparian vegetation. A monitoring plan should be included in any revegetation project. For projects that disturb large areas of riparian vegetation, performance bonds should also be required. Each of these measures is discussed more fully below.

For large projects with high-quality riparian habitat that require extensive access from the shoreline for construction, consider the short-term impacts of work performed in the channel rather than removing high-quality riparian habitat that would be a long-term impact due to the size and age of the stand.

Consider establishing buffers and setbacks that protect the functions of the riparian system and its contribution to ecosystem. The term "buffer" is often loosely used as a synonym for riparian area. However, the term buffer is typically applied in a specific management context to denote an area set aside and managed to protect a natural environment from the effects of surrounding land-use or human activities (May 2003; Knutson and Naef 1997). Depending on the context, buffers may be designed to perform a specific function or set of functions, such as filtering pollutants or providing shade (May 2003). The use of the term "buffer" in this white paper and the recommendations therein are directed to protect the area ("riparian protection area") needed for the ecological functions of riverine, lacustrine, and nearshore marine habitats.

Establishing buffer areas is an important regulatory tool both to keep development activities in this habitat to a minimum, and (for developed or redeveloping sites) to trigger mitigation sequencing to deal with project impacts on riparian vegetation. May (2003) provides a review of riparian functions as a factor of buffer width. Table 11-1 provides a summary from the scientific literature of how different riparian habitat widths protect function. As indicated in May (2003), there is no consensus in the literature recommending a single buffer width for a particular function or to accommodate all functions. Knutson and Naef (1997) resolved the variability in the literature by averaging effective buffers widths reported for specific riparian functions. Table 11-2 illustrates the results of the Knutson and Naef (1997) literature review and shows that for streams, a buffer width of 147 feet is effective in providing five of the seven riparian functions including: sediment filtration, erosion control, pollutant removal, LWD, and water temperature protection.

Table 11-1. Riparian buffer functions and appropriate widths identified by May (2003).

Riparian Function	Range of Effective Buffer Widths (feet)	Minimum Recommended Widths (feet)	Notes on Function
Sediment removal/erosion control	26 – 600	98	For 80% sediment removal
Pollutant removal	13 – 860	98	For 80% nutrient removal
LWD recruitment	33 – 328	164	1 SPTH based on long-term natural levels
Water temperature	36 – 141	98	Based on adequate shade
Wildlife habitat	33 – 984	328	Coverage not inclusive
Microclimate	148 – 656	328	Optimum long-term support

SPTH = site potential tree height.

Table 11-2. Riparian functions and appropriate widths identified by Knutson and Naef (1997).

Function	Range of Effective Buffer Widths (feet)	Average of Reported Widths (feet)
Sediment filtration	26 – 300	138
Erosion control	100 – 125	112
Pollutant removal	13 – 600	78
LWD recruitment	100 – 200	147
Water temperature protection	35 – 151	90
Wildlife habitat	25 – 984	287
Microclimate	200 – 525	412

Additionally, to protect and restore riparian habitat functions, management strategies should:

- Prohibit the removal or disturbance of riparian vegetation for any areas subject to erosion hazard.
- Fill data gaps through research and documentation of successful and failed riparian protection and revegetation strategies to develop effective policies that protect riparian functions that are important to HCP species.
- Establish buffers and setbacks that protect the functions of the riparian system and its contribution to ecological functions.
- Maintain and restore riparian vegetation to protect human health and safety.
- If the project removes vegetation, require that the project proponent save the large trees and root wads to place strategically in either this aquatic habitat or another restoration project in the region.
- Where riparian vegetation has been removed, isolate disturbed areas from aquatic resources using erosion control features until disturbed areas are stabilized.
- Incorporate all ecological functions into the riparian management strategy.
- Develop financial incentives for conservation programs.
- Increase public education and outreach to educate the public and decision-makers on the outcomes of project actions and decisions.

11.1.2.1 Revegetation Design

To protect habitat and ecological functions for HCP species, revegetation plans should only include native species endemic to the location of the project. The proximity of the vegetation to the aquatic habitat and the size of the vegetation should be such that it can restore the ecological benefits, such as temperature regulation and allochthonous inputs.

11.1.2.2 Monitoring Plan

Pursuant to WAC 220-110, revegetation should be monitored annually for 3 years to ensure 100 percent survival of all plantings at the end of 1 year and 80 percent survival by the end of the 3-year monitoring period. Monitoring data should be provided to permitting agencies in detailed annual monitoring reports. After 3 years, monitoring and reporting should be completed every other year or every third year. In addition, any specific conditions provided by the U.S. Army Corps of Engineers (for project permits) or NOAA Fisheries and USFWS (for ESA Section 7 compliance) must be implemented.

11.1.2.3 Insurance

Require performance bonds to cover projects that disturb large areas of riparian vegetation.

11.1.3 Aquatic Vegetation Modifications

As illustrated in Section 7 (*Direct and Indirect Impacts*), flow control structures will impact aquatic vegetation through altered autochthonous production, habitat complexity, and nutrient cycling. These impacts can be minimized or prevented altogether, for example, by locating the facility in an area that is currently devoid of native aquatic vegetation or in an area that will minimize the potential impacts. In addition, construction of flow control structures at a time of year when aquatic vegetation biomass is at a minimum is recommended.

To protect and restore aquatic habitat functions, management strategies and development of shoreline regulations should:

- Avoid or minimize the removal or disturbance of aquatic vegetation.
- Manage equipment operations and establish no-construction or no-vessel activity buffers around existing aquatic vegetation to protect this habitat and its contribution to ecological functions.
- Require the control of turbidity during construction and operation of the facility to prevent suffocation or excessive shading of plants.
- Site and design flow control structures in deeper water to minimize shading and physical impacts on aquatic vegetation.

- Place the potential shade-casting structures perpendicular to the arc of the sun (i.e., north–south placement) to maximize transmission of light under the structure.

- Any walkways should be 100 percent grated; floats and docks should be at least 60 percent grating.

- Orient grating to maximize transmission of light under the structure.

11.1.4 Lost-Opportunity Impacts

The hydraulic and geomorphic modifications induced by many flow control structures, particularly dams, weirs and tide gates, can result in lost-opportunity impacts. Mitigation for lost opportunity requires mitigation for channel processes affected by a project. In some situations, off-site mitigation may be the only option (WDFW 2003). According to WDFW (2003), the concept of mitigation for lost opportunity should only be applied when consistent, acceptable assessment methods or site-specific information is available. More detailed information on mitigation for lost-opportunity is provided in WDFW (2003).

11.2 Dams

Dams severely alter natural rivers systems in many ways including physically blocking the movement of migrating species, altering the natural flow regime, and reducing suitable habitats. Mitigation of these impacts can be divided into three general groups: (1) actions to improve fish passage, (2) actions to restore natural flow regime, and (3) actions to reduce water quality impacts. In addition, certain actions can be taken during the construction phase of dam projects to minimize impacts from construction which are listed in Section 11.1.1 (*Construction and Maintenance Activities*). Finally, the special case of dam removal will often serve to reverse or greatly minimize impacts from dam projects in the long term. These mitigation strategies are briefly described below.

11.2.1 Fish Passage

To minimize migratory impacts from dams, adequate fish passage structures are required that allow a majority of fish to reach upstream and downstream habitats. For example, Webber et al. (2007) concluded that the design of dams and fish barriers should have fast and slow portions to increase migration over these structures. In laboratory studies, the authors demonstrated that white sturgeon attempt to pass barriers with short bursts, followed by a resting period. Therefore, design of fish barriers (e.g., weirs, dams, step-pools) should have fast sections 2.76–8.27 ft/sec (0.84–2.52 m/sec), followed by slower sections 1.64–2.23 ft/sec (0.5–0.68 m/sec) for recovery (Webber et al. 2007). Information on optimal swimming velocities, height restrictions, diurnal migration patterns, and behavior at passage facilities for HCP species will be necessary to optimize fish passage in the presence of dams. For a detailed analysis of the impacts of fish passage structures and their mitigation strategies, refer to the Fish Passage white paper (Herrera 2007a).

11.2.2 Flow Regime

Numerous studies have concluded that in order to maintain the ecological integrity of riverine environments in the presence of dams, some return to a natural flow regime is needed (Bednarek 2001). A return to the natural flow regime will maintain habitat complexity and connectivity, limit impacts from altered sediment transport and substrate composition, and improve species diversity. These are sometimes referred to as environmental flows (Chester and Norris 2006). In the Grand Canyon, attempts to remediate sediment movement by prescribed flooding or higher (elevation) releases of water through dams have taken place. Collier et al. (1997) documented that incised beaches and sand bars downstream of Glen Canyon dam were somewhat restored during these “flood” events. However, beaches and sandbars still suffered from a reduction in sediment supply.

Biodiversity is best protected where dam operation emulates a natural system. Food webs require variable flow regime and floodplain inundation (Power et al. 1996). Environmental flows used in Australia showed that macroinvertebrate communities were similar to those of unregulated flows in the region (Chester and Norris 2006). In addition, flow releases that simulate variable flows have been observed to improve the diversity of warmwater fish assemblages (Travnichek et al. 1995). On the Tallapoosa River (Alabama), the relative abundance of species classified as fluvial specialists increased from below 40 to more than 80 percent after initiating a more variable flow regime.

11.2.3 Water Quality

The primary impacts from dams on water quality include altered temperatures and altered dissolved oxygen concentrations. These modifications can be minimized if water releases from the reservoir can occur at multiple depths (Bednarek 2001). This mitigation practice will vary depending on the local conditions, as well as on what species are present; therefore, this practice should be analyzed on a case-by-case basis. In some cases, multiple-depth flow releases will solve these water quality problems; in other cases, they will not (Bednarek 2001).

11.2.4 Dam Removal

Dam removal is the best way to reestablish thermal regimes and natural sediment transport, restore habitat complexity, and minimize water quality changes. Dam removals are becoming more common as facilities are applying to renew licenses because, in some cases, dam removal is a more economical or safer option (Bednarek 2001). Dam removal, in general, will restore natural sediment transport in the system by increasing habitat diversity in the former impoundments (Bednarek 2001) and replenishing coastal systems where beach erosion has proliferated (DOI 1995). Recently, eulachon have been observed in the Elwha River (Washington), and dam removal will likely increase the availability of sand and gravel sizes required for these fish to spawn (Shaffer et al. 2007). Dam removal will allow organisms to migrate freely, reduce delays in migration, and reduce mortality caused by fish passage structures (Travnichek et al. 1993).

One significant environmental concern from dam removal projects is the release of stored sediment from the former impoundment. Stored sediments may cause increases in downstream sediment transport and turbidity; however, these increases will be a short-term impact while the river transitions back to a free-flowing system. Factors influencing the duration of impact from sediment releases from a dam removal include: (1) the length of time dam was present, (2) velocity and gradient of river, and (3) removal techniques (Bednarek 2001). The frequency of storms after removal is also important. The downstream effects from sediment releases can be on the order of days (Winter 1990) to many years. In some cases, sediment release will be equivalent to a periodic storm event (Winter 1990). Along with increases in turbidity, there is the potential for contamination arising from pollutants that are adsorbed onto sediment particulates. Pollutant contamination can be reduced by conducting a preremoval evaluation of sediments or dredging, and by conducting a slow drawdown of the reservoir prior to dam removal (Bednarek 2001).

11.3 Weirs

Weirs are similar to dams but are generally smaller in scale. As a result, mitigation activities associated with weirs are identical to those described for dams. See Section 11.2 (*Dams*) for details.

11.4 Dikes and Levees

Breaching of dikes and levees has been used to reconnect channel and floodplain habitats, with several documented benefits. After breaching levees on the Consumes River (California), floodplain geomorphology became more complex, with changes in topography, woody debris recruitment, and vegetation (Florsheim and Mount 2002). In addition, restored connectivity has been shown to enhance nutrient cycling by reducing nitrate loading downstream (Sheibley et al. 2006). Finally, levee breaches can influence algal dynamics and overall water quality of the restored floodplain (Ahearn et al. 2006).

Erosion and failure of levees may be reduced through planting vegetation. Conversely, vegetation removal is often encouraged on levees to provide access for inspection, fight flooding, reduce rodent burrowing, and to prevent root-induced water removal (Bolton and Shellberg 2001). However, this study also noted that grass and vegetation actually stabilize these structures, similar to vegetated stream banks. In addition, grass coverage on levees will cause a more even wetting and drying of the structures through transpiration, which will lessen cracking and failure from uneven drying after flood events. Finally, taller vegetation may shade levees and reduce the cracking of earthen levees from extreme heat.

Where possible, dike and levee projects should be designed to retain as much natural hydraulic and geomorphic features as possible. This can be achieved by increasing the distance between the levees to allow channels to naturally meander, incorporating meanders into the channelization project, minimizing the reach length where levees are constructed, or creating artificial side channels (Bolton and Shellberg 2001). The creation of artificial side channels will simulate a low-flow channel; when flooding occurs, water spills out into the "floodplain" and creates side channel and side pool habitats. Finally, levee projects can be conducted so that in-channel (e.g., pools, riffles) features are preserved (Bolton and Shellberg 2001). This can easily be achieved by not dredging the channel after the levee or dike is constructed. As with all of these mitigation strategies, their feasibility depends on several site-specific factors, including the purpose of the project, the size of the project area, cost, and safety.

For construction activities associated with dikes and levees, mitigation procedures are addressed in general terms in Section 11.1.1 (*Construction and Maintenance Activities*).

11.5 Outfalls

Hydraulic and geomorphic modifications associated with outfalls can be eliminated with a design that minimizes alterations to the physical environment surrounding the outlet. A few recommendations are:

- Locate all outfall infrastructure below-grade in areas where sediment transport is significant.
- Place submerged outfall outlets below the closure depth or light penetration depth, whichever is greater.
- Where possible, avoid discharges that are significantly different in density, temperature, salinity, and turbidity from the receiving water.
- Minimize the flow velocities of the discharged fluid. If the flow rates are expected to significantly alter the circulation or geomorphology in the vicinity of the outlet, hydrodynamic modeling should be performed to assess and limit the area of impact.
- To avoid scour associated with large discharge velocities, site the outfall outlet in an area of pre-existing immobile substrate, where possible.
- Screen the outlet to prevent fish entrainment into the outfall piping.
- Exposed outfalls should be sited such that they do not protrude or disrupt sediment transport. Where possible, placement of the outlet should be approved by a licensed geologist.

Where hydraulic and geomorphic modifications are unavoidable, mitigation of such effects is necessary. This could include the routing of sediment around the geomorphic disruption, as in the case of weir jetties (see Shoreline Modifications white paper [(Herrera 2007f)] for details). Monitoring plans associated with submerged outfalls should also include ongoing inspections of the outlet infrastructure for the presence of invasive species.

One of the most significant impacts from outfall projects is the alteration of water quality in receiving waters. These impacts can be minimized by ensuring that the contaminant load in the effluent has been reduced to the greatest extent possible, and by locating outfalls in marine areas where dilution and flushing are maximized (Williams and Thom 2001). In riverine environments, establishing a mixing zone will lower the effects downstream; because sediments are associated with many types of pollutants (Murakami and Takeishi 1977), reducing the amount of sediment in the outfall discharge is desirable.

For construction activities associated with outfalls, mitigation procedures are addressed in general terms in Section 11.1.1 (*Construction and Maintenance Activities*).

11.6 Intakes and Diversions

The primary hydraulic and geomorphic alterations associated with intakes and diversions are related to the piping infrastructure for these systems. See Section 11.5 (*Outfalls*) for design and maintenance recommendations. In addition, the most common issue related to intakes and diversions is the entrainment of fish and invertebrates. This impact is mitigated by using fish screens and is addressed in a separate white paper (Herrera 2007b).

Alteration of the amount of water removed and the timing of water removals can minimize impacts related to these structures. For example, a study of downstream drifting shrimp larvae showed that a large percentage of the larvae can be entrained in water intakes, with a mortality of 42 percent and almost 100 percent removed from water column during low flows (Benstead et al. 1999). However, the authors showed that most drift took place at night. When the intake was turned off for 5 hours at night, mortality was reduced to 11–20 percent (Benstead et al. 1999). This study demonstrates that knowing the migration and behavior patterns of HCP species will allow managers to minimize the impacts from flow control structures such as water intakes and diversions. In addition, Miller et al. (2007) discuss that to minimize impacts from diversions on macroinvertebrate communities, diversions should preserve environmental conditions as much as possible.

In terms of construction activities associated with intakes and diversions, mitigation procedures have been addressed in general terms in Section 11.1.1 (*Construction and Maintenance Activities*).

11.7 Tide Gates

As with the presence of a dam, tide gates can significantly alter the migration of aquatic organisms and change the natural flow regime. The less time a tide gate is closed, the less likely the impacts on HCP species will be. The type of tide gate and the materials used for its construction can influence how long the gate remains open during the day. Tide gate design is summarized in Giannico and Souder (2005), and improvements for fish passage are described in Charland (1998). In general, tide boxes with side-hinged gates result in lower velocities required to open the gate compared to top-hinged gates because less force is needed to open them. Also, gates constructed of lighter aluminum need less water to open than heavier steel or cast iron gates of comparable size (Giannico and Souder 2005). In addition, side-hinged gates open slower such that they also reduce bubbling, turbulence, and scour (Giannico and Souder 2005). In addition, if information is known about the local behavior and migration patterns of HCP fish or other species, tide boxes may be manually opened to maximize passage during migration and other high-use periods.

In terms of construction activities associated with tide gates, mitigation procedures are addressed in general terms in Section 11.1.1 (*Construction and Maintenance Activities*).

12.0 References

Aarestrup, K., and A. Koed. 2003. Survival of Migrating Sea Trout (*Salmo Trutta*) and Atlantic Salmon (*Salmo Salar)* Smolts Negotiating Weirs in Small Danish Rivers. *Ecology of Freshwater Fish* 12(3): 169-176.

Abbe, T.B., and D.R. Montgomery. 1996. Large Woody Debris Jams, Channel Hydraulics and Habitat Formation in Large Rivers. *Regulated Rivers-Research & Management* 12(2-3): 201-221.

Abbe, T.B., and D.R. Montgomery. 2003. Patterns and Processes of Wood Debris Accumulation in the Queets River Basin, Washington. *Geomorphology* 51(1-3): 81-107.

Abdelrhman, M.A. 2003. Effect of Eelgrass *Zostera Marina* Canopies on Flow and Transport. *Marine Ecology-Progress Series* 248: 67-83.

Adams, P.B., and J.E. Hardwick. 1992. Lingcod. In *California's Living Marine Resources and Their Utilization*, edited by W.S. Leet, C.M. Dewees and C.W. Haugen. Davis, California: California Sea Grant College Program.

Adams, P.B., C.B. Grimes, J.E. Hightower, S.T. Lindley, and M.L. Moser. 2002. Status Review for North American Green Sturgeon, *Acipenser Mediorstris*. National Marine Fisheries Service, Southwest Fisheries Science Center.

AFS and SER. 2000. Review of the 29 April 1999 “Forests and Fish Report” and of Associated “Draft Emergency Forest Practice Rules.” Northwest Chapter of the Society for Ecological Restoration.

Ahearn, D.S., J.H. Viers, J.F. Mount, and R.A. Dahlgren. 2006. Priming the Productivity Pump: Flood Pulse Driven Trends in Suspended Algal Biomass Distribution across a Restored Floodplain. *Freshwater Biology* 51: 1417-1433.

Ahearn, D.S., R.W. Sheibley, and R.A. Dahlgren. 2005. Effects of River Regulation on Water Quality in the Lower Mokelumne River, California. *River Research and Applications* 21(6): 651-670.

Ainslie, B.J., J.R. Post, and A.J. Paul. 1998. Effects of Pulsed and Continuous DC Electrofishing on Juvenile Rainbow Trout. *North American Journal of Fisheries Management* 18 (4): 905-918.

Airoldi, L., and S.J. Hawkins. 2007. Negative Effects of Sediment Deposition on Grazing Activity and Survival of the Limpet *Patella Vulgata*. *Marine Ecology-Progress Series* 332: 235-240.

Aksnes, D.L., and A.C.W. Utne. 1997. A Revised Model of Visual Range in Fish. *Sarsia* 82(2): 137-147.

Albers, W.D., and P.J. Anderson. 1985. Diet of Pacific Cod, *Gadus Macrocephalus*, and Predation on the Northern Pink Shrimp, *Pandalus Borealis*, in Pavlof Bay, Alaska. *Fishery Bulletin* 83: 601-10.

Amoser, S., and F. Ladich. 2005. Are Hearing Sensitivities of Freshwater Fish Adapted to the Ambient Noise in Their Habitats? *The Journal of Experimental Biology* 208: 3533-3542.

Angermeier, P.L., and J.R. Karr. 1984. Relationships between Woody Debris and Fish Habitat in a Small Warmwater Stream. *Transaction of the American Fisheries Society* 113: 716-726.

Anisfeld, S.C., and G. Benoit. 1997. Impacts of Flow Restrictions on Salt Marshes: An Instance of Acidification. *Environmental Science and Technology* 31: 1650-1657.

Anisfeld, S.C., M.J. Tobin, and B. Gaboury. 1999. Sedimentation Rates in Flow-Restricted and Restored Salt Marshes in Long Island Sound. *Estuaries and Coasts* 22(2A): 231-244.

Armstrong, D.A., B.G. Stevens, and J.E. Hoeman. 1982. Distribution and Abundance of Dungeness Crab and Crangon Shrimp and Dredging-Related Mortality of Invertebrates and Fish in Grays Harbor, Washington. DACW67-80-C-0086. Seattle, Washington: School of Fisheries, University of Washington.

Assani, A.A., and F. Petit. 2004. Impact of Hydroelectric Power Releases on the Morphology and Sedimentology of the Bed of the Warche River (Belgium). *Earth Surface Processes and Landforms* 29(2): 133-143.

Au, D.W.T., C.A. Pollino, R.S.S. Wu, P.K.S. Shin, S.T.F. Lau, and J.Y.M. Tang. 2004. Chronic Effects of Suspended Solids on Gill Structure, Osmoregulation, Growth, and Triiodothyronine in Juvenile Green Grouper *Epinephelus Coioides*. *Marine Ecology-Progress Series* 266: 255-264.

Babanin, A.V. 2006. On a Wave-Induced Turbulence and a Wave-Mixed Upper Ocean Layer. *Geophysical Research Letters* 33(20).

Bacchiocchi, F., and L. Airoldi. 2003. Distribution and Dynamics of Epibiota on Hard Structures for Coastal Protection. *Estuarine Coastal and Shelf Science* 56(5-6): 1157-1166.

Bailey, K.M. 1982. The Early Life History of the Pacific Hake, *Merluccius Productus*. *Fishery Bulletin* 80: 589-598.

Bailey, K.M., T.J. Quinn, P. Bentzen, and W.S. Grant. 1999. Population Structure and Dynamics of Walleye Pollock, *Theragra Chalcogramma*. *Advances in Marine Biology* 37: 179-255.

Baker, C.F. 2003. Effect of Fall Height and Notch Shape on the Passage of Inanga (*Galaxias Maculatus*) and Common Bullies (*Gobiomorphus Cotidianus*) over an Experimental Weir. *New Zealand Journal of Marine and Freshwater Research* 37(2): 283-290.

Baker, P. 1995. Review of Ecology and Fishery of the Olympia Oyster, *Ostrea Lurida* with Annotated Bibliography. *Journal of Shellfish Research* 14(2): 501-518.

Baldwin, D.H., J.F. Sandahl, J.S. Labenia, and N.L. Scholz. 2003. Sublethal Effects of Copper on Coho Salmon: Impacts on Nonoverlapping Receptor Pathways in the Peripheral Olfactory Nervous System. *Environmental Toxicology and Chemistry* 22(10): 2266-2274.

Banner, A., and M. Hyatt. 1973. Effects of Noise on Eggs and Larvae of Two Estuarine Fishes. *Transaction of the American Fisheries Society* 102: 134-136.

Barber, L.B., S.F. Murphy, P.L. Verplanck, M.W. Sandstrom, H.E. Taylor, and E.T. Furlong. 2006. Chemical Loading into Surface Water Along a Hydrological, Biogeochemical, and Land Use Gradient: A Holistic Watershed Approach. *Environmental Science & Technology* 40(2): 475-486.

Bargmann, G.C. 1980. Studies on Pacific Cod in Agate Pass, Washington. Washington Department of Fish and Wildlife.

Bartholow, J.M. 2002. Estimating Cumulative Effects of Clearcutting on Stream Temperatures. Available at: http://smig.usgs.gov/SMIG/features_0902/clearcut.html.

Barton, D.R., W.D. Taylor, and R.M. Biette. 1985. Dimensions of Riparian Buffer Strips Required to Maintain Trout Habitat in Southern Ontario Streams. *North American Journal of Fisheries Management* 5: 364-378.

Bash, J., C. Berman, and S. Bolton. 2001. Effects of Turbidity and Suspended Solids on Salmonids. Washington State Transportation Center.

Baxter, C.V., and F.R. Hauer. 2000. Geomorphology, Hyporheic Exchange, and Selection of Spawning Habitat by Bull Trout (*Salvelinus Confluentus*). *Canadian Journal of Fisheries and Aquatic Sciences* 57(7): 1470-1481.

Baxter, J.S., and J.D. McPhail. 1999. The Influence of Redd Site Selection, Groundwater Upwelling, and over-Winter Incubation Temperature on Survival of Bull Trout (*Salvelinus Confluentus*) from Egg to Alevin. *Canadian Journal of Zoology-Revue Canadienne De Zoologie* 77(8): 1233-1239.

Baylar, A., and T. Bagatur. 2000. Aeration Performance of Weirs. *Water SA* 26(4): 521-526.

Beacham, T.D., and C.B. Murray. 1990. Temperature, Egg Size, and Development of Embryos and Alevins of 5 Species of Pacific Salmon - a Comparative-Analysis. *Transactions of the American Fisheries Society* 119(6): 927-945.

Beamer, E., Aundrea McBride, Correigh Greene, Rich Henderson, Greg Hood, Karen Wolf, Kim Larsen, Casey Rice, and Kurt Fresh. 2005. Delta and Nearshore Restoration for the Recovery of Wild Skagit River Chinook Salmon: Linking Estuary Restoration to Wild Chinook Salmon Populations. LaConner, Washington: Skagit River System Cooperative.

Beamer, E.M., and R.A. Henderson. 1998. Juvenile Salmonid Use of Natural and Hydromodified Stream Bank Habitat in the Mainstem Skagit River, Northwest Washington. Prepared for United States Army Corps of Engineers, Seattle District, Environmental Resources Section, Seattle, Washington. September 1998.

Bednarek, A.T. 2001. Undamming Rivers: A Review of the Ecological Impacts of Dam Removal. *Environmental Management* 27(6): 803-814.

Beeman, J.W., D.W. Rondorf, and M.E. Tilson. 1994. Assessing Smoltification of Juvenile Spring Chinook Salmon (*Oncorhynchus-Tshawytscha*) Using Changes in Body Morphology. *Canadian Journal of Fisheries and Aquatic Sciences* 51(4): 836-844.

Bennett, D.H., W.P. Connor, and C.A. Eaton. 2003. Substrate Composition and Emergence Success of Fall Chinook Salmon in the Snake River. *Northwest Science* 77(2): 93-99.

Benstead, J.P., J.G. March, C.M. Pringle, and F.N. Scatena. 1999. Effects of a Low-Head Dam and Water Abstraction on Migratory Tropical Stream Biota. *Ecological Applications* 9(2): 656-668.

Berg, L., and T.G. Northcote. 1985. Changes in Territorial, Gill-Flaring, and Feeding-Behavior in Juvenile Coho Salmon (*Oncorhynchus-Kisutch)* Following Short-Term Pulses of Suspended Sediment. *Canadian Journal of Fisheries and Aquatic Sciences* 42(8): 1410-1417.

Berry, W., N. Rubenstein, B. Melzian, and B. Hill. 2003. The Biological Effects of Suspended and Bedded Sediments (SABS) in Aquatic Systems: A Review. Narragansett, Rhode Island: U.S. Environmental Protection Agency, Office of Research and Development.

Beschta, R.L. 1991. Stream Habitat Management for Fish in the Northwestern United States: The Role of Riparian Vegetation. *American Fisheries Society Symposium* 10: 53-58.

Beschta, R.L. 1997. Riparian Shade and Stream Temperature: An Alternative Perspective. *Rangelands* 19(2): 25-28.

Beschta, R.L., and R.L. Taylor. 1988. Stream Temperature Increases and Land-Use in a Forested Oregon Watershed. *Water Resources Bulletin* 24(1): 19-25.

Beschta, R.L., R.E. Bilby, G.W. Brown, L.B. Holtby, and T.D. Hofstra. 1988. Stream Temperature and Aquatic Habitat: Fishery and Forestry Interactions. In *Streamside Management: Forestry and Fishery Interactions: Contribution No. 57*. Seattle, Washington: University of Washington, Institute of Forest Resources. pp. 191-232.

Bilby, R.E. 1984. Post-Logging Removal of Woody Debris May Affect Stream Channel Stability. *Journal of Forestry* 82(10): 609-613.

Bilby, R.E., and P.A. Bisson. 1992. Allochthonous Versus Autochthonous Organic-Matter Contributions to the Trophic Support of Fish Populations in Clear-Cut and Old-Growth Forested Streams. *Canadian Journal of Fisheries and Aquatic Sciences* 49(3): 540-551.

Bilby, R.E., and P.A. Bisson. 1998. Function and Distribution of Large Woody Debris. In *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*, edited by R.J. Naiman and R.E. Bilby. Springer-Verlag, New York. pp. 324-347.

Bjornn, T.C., and D.W. Reiser. 1991. Habitat Requirements of Salmonids in Streams. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*, edited by W.R. Meehan. Bethesda, Maryland: American Fisheries Society Special Publication 19. pp. 83-138.

Blaber, S.J.M., D.P. Cyrus, J.J. Albaret, C.V. Ching, J.W. Day, M. Elliott, M.S. Fonseca, D.E. Hoss, J. Orensanz, I.C. Potter, and W. Silvert. 2000. Effects of Fishing on the Structure and Functioning of Estuarine and Nearshore Ecosystems. *ICES Journal of Marine Science* 57(3): 590-602.

Black, M.C., D.S. Millsap, and J.F. Mccarthy. 1991. Effects of Acute Temperature-Change on Respiration and Toxicant Uptake by Rainbow-Trout, *Salmo-Gairdneri* (*Richardson*). *Physiological Zoology* 64(1): 145-168.

Blaxter, J.H.S., J.A.B. Gray, and E.J. Denton. 1981. Sound and Startle Responses in Herring Shoals. *Journal of Marine Biology Associated with U.K.* 6: 851-869.

Blomqvist, S., A. Gunnars, and R. Elmgren. 2004. Why the Limiting Nutrient Differs between Temperate Coastal Seas and Freshwater Lakes: A Matter of Salt. *Limnology and Oceanography* 49(6): 2236-2241.

Boehlert, G.W. 1980. Size Composition, Age Composition, and Growth of Canary Rockfish, *Sebastes Pinniger*, and Splitnose Rockfish, *S. Diploproa*, from the 1977 Rockfish Survey. *Marine Fisheries Review* 42: 57-63.

Boehlert, G.W., and R.F. Kappenman. 1980. Variation of Growth with Latitude in Two Species of Rockfish (*Sebastes Pinniger* and *S. Diploproa*) from the Northeast Pacific Ocean. *Marine Ecology-Progress Series* 3: 1-10.

Boehlert, G.W., M.M. Yoklavich, and D.B. Chelton. 1989. Time Series of Growth in the Genus *Sebastes* from the Northeast Pacific Ocean. *Fishery Bulletin* 87: 791-806.

Bolton, S., and J. Shellberg. 2001. Ecological Issues in Floodplains and Riparian Corridors. Prepared for Washington Department of Fish and Wildlife; Washington Department of Ecology; Washington Department of Transportation by University of Washington, Center for Streamside Studies, Seattle, Washington. July 11, 2001.

Bostrom, C., and J. Mattila. 1999. The Relative Importance of Food and Shelter for Seagrass-Associated Invertebrates: A Latitudinal Comparison of Habitat Choice by Isopod Grazers. *Oecologia* 120(1): 162-170.

Bottom, D.L., K.K. Jones, T.J. Cornwell, A. Gray, and C.A. Simenstad. 2005. Patterns of Chinook Salmon Migration and Residency in the Salmon River Estuary (Oregon). *Estuarine Coastal and Shelf Science* 64(1): 79-93.

Bowen, K.L., N.K. Kaushik, and A.M. Gordon. 1998. Macroinvertebrate Communities and Biofilm Chlorophyll on Woody Debris in Two Canadian Oligotrophic Lakes. *Archiv Fur Hydrobiologie* 141(3): 257-281.

Bowman, M.F., and R.C. Bailey. 1998. Upper pH Tolerance Limit of the Zebra Mussel (*Dreissena Polymorpha*). *Canadian Journal of Zoology-Revue Canadienne De Zoologie* 76(11): 2119-2123.

Box, J.B., D. Wolf, J. Howard, C. O'Brien, D. Nez, and D. Close. 2003. The Distribution and Status of Freshwater Mussels in the Umatilla River System. Portland, Oregon: Bonneville Power Administration.

Bramblett, R.G., M.D. Bryant, B.E. Wright, and R.G. White. 2002. Seasonal Use of Small Tributary and Main-Stem Habitats by Juvenile Steelhead, Coho Salmon, and Dolly Varden in a Southeastern Alaska Drainage Basin. *Transactions of the American Fisheries Society* 131(3): 498-506.

Brannon, E.L., M.S. Powell, T.P. Quinn, and A. Talbot. 2004. Population Structure of Columbia River Basin Chinook Salmon and Steelhead Trout. *Reviews in Fisheries Science* 12(2-3): 99-232.

Brennan, J. 2004. Riparian Functions and the Development of Management Actions in Marine Nearshore Ecosystems. In *Proceedings of the DFO/PSAT Sponsored Marine Riparian Experts Workshop, Tsawwassen, BC, February 17-18, 2004*, edited by J.P. Lemieux, J.S. Brennan, M. Farrell, C.D. Levings and D. Myers. Canadian Manuscript Report of the Fisheries and Aquatic Sciences No. 2680.

Brennan, J.S., and H. Culverwell. 2004. Marine Riparian: An Assessment of Riparian Functions in Marine Ecosystems. Seattle, Washington: Washington Sea Grant Program.

Brennan, J.S., K.F. Higgins, J.R. Cordell, and V.A. Stamatiou. 2004. Juvenile Salmon Composition, Timing Distribution, and Diet in Marine Nearshore Waters of Central Puget Sound in 2001-2002. Seattle, Washington: King County Department of Natural Resources and Parks.

Bricelj, V.M., and R.E. Malouf. 1984. Influence of Algal and Suspended Sediment Concentrations on the Feeding Physiology of the Hard Clam *Mercenaria-Mercenaria*. *Marine Biology* 84(2): 155-165.

Bricelj, V.M., R.E. Malouf, and C. Dequillfeldt. 1984. Growth of Juvenile *Mercenaria-Mercenaria* and the Effect of Resuspended Bottom Sediments. *Marine Biology* 84(2): 167-173.

Brosofske, K.D., J. Chen, R.J. Naiman, and J.F. Franklin. 1997. Harvesting Effects on Microclimatic Gradients from Small Streams to Uplands in Western Washington. *Ecological Applications* 7(4): 1188-1200.

Brown, G.W. 1970. Predicting Effect of Clearcutting on Stream Temperature. *Journal of Soil and Water Conservation* 25(1): 11-13.

Brown, R.S., D.R. Geist, and the Yakama Nation. 2002. Determination of Swimming Speeds and Energetic Demands of Upriver Migrating Fall Chinook Salmon (*Oncorhynchus Tshawytscha*) in the Klickitat River, Washington. Pacific Northwest Laboratory; Bonneville Power Administration.

Bryant, M.D., T. Gomi, and J.J. Piccolo. 2007. Structures Linking Physical and Biological Processes in Headwater Streams of the Maybeso Watershed, Southeast Alaska. *Forest Science* 53(2): 371-383.

Buck, E.H. 1995. Acoustic Thermometry of Ocean Climate, Marine Mammal Issues. Report No. 95-603 ENR. Washington DC: Congressional Research Service, Library of Congress.

Buell, J. 1992. Fish Entrainment Monitoring of the Western-Pacific Dredge *R.W. Lofgren* During Operations Outside the Preferred Work Period. Portland, Oregon.

Buer, K.Y., J.N. Eaves, R.G. Scott, and J.R. McMillan. 1984. Basin Changes Affecting Salmon Habitat in the Sacramento River. *Pacific Northwest stream habitat management workshop*, Arcata, California.

Bulleri, F., and M.G. Chapman. 2004. Intertidal Assemblages on Artificial and Natural Habitats in Marinas on the North-West Coast of Italy. *Marine Biology* 145(2): 381-391.

Bunn, S.E., and A.H. Arthington. 2002. Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. *Environmental Management* 30(4): 492-507.

Burgess, W.C., and S.B. Blackwell. 2003. Acoustic Monitoring of Barrier Wall Installation at the Former Rhône-Poulenc Site, Tukwila, Washington. Tukwila, Washington: Greenridge Sciences, Inc.

Burner, L.C., and T.A. Rien. 2002. Incidence of White Sturgeon Deformities in Two Reaches of the Columbia River. *California Fish and Game* 88(2): 57-67.

Busby, M.S., and R.A. Barnhart. 1995. Potential Food Sources and Feeding Ecology of Juvenile Fall Chinook Salmon in California's Mattole River Lagoon. *California Fish and Game* 81(4): 133-146.

Busby, P.J., T.C. Wainwright, G.J. Bryant, L.J. Lierheimer, R.S. Waples, F.W. Waknitz, and I.V. Lagomarsino. 1996. Status Review of West Coast Steelhead from Washington, Idaho, Oregon, and California. Seattle, Washington: National Oceanic and Atmospheric Administration.

Byrnes, M.R., and M.W. Hiland. 1995. Large-Scale Sediment Transport Patterns on the Continental-Shelf and Influence on Shoreline Response - St-Andrew-Sound, Georgia to Nassau-Sound, Florida, USA. *Marine Geology* 126(1-4): 19-43.

Byrnes, M.R., R.M. Hammert, T.D. Thibaut, and D.B. Snyder. 2004. Effects of Sand Mining on Physical Processes and Biological Communities Offshore New Jersey, USA. *Journal of Coastal Research* 20(1): 25-43.

Cake, E.W.J. 1983. Habitat Suitability Index Models: Gulf of Mexico American Oyster. U.S. Department of Interior, Fish and Wildlife Service, Washington D.C.

Camargo, J.A., and N.J. Voelz. 1998. Biotic and Abiotic Changes Along the Recovery Gradient of Two Impounded Rivers with Different Impoundment Use. *Environmental Monitoring and Assessment* 50: 143-158.

Camargo, J.A., K. Alonso, and M. de la Puente. 2005. Eutrophication Downstream from Small Reservoirs in Mountain Rivers of Central Spain. *Water Research* 39(14): 3376-3384.

Cardoso, P.G., D. Raffaelli, and M.A. Pardal. 2007. Seagrass Beds and Intertidal Invertebrates: An Experimental Test of the Role of Habitat Structure. *Hydrobiologia* 575: 221-230.

Carls, M.G., J.E. Hose, R.E. Thomas, and S.D. Rice. 2000. Exposure of Pacific Herring to Weathered Crude Oil: Assessing Effects on Ova. *Environmental Toxicology and Chemistry* 19(6): 1649-1659.

Carlson, T.J., D.A. Woodruff, G.E. Johnson, N.P. Kohn, G.R. Plosky, M.A. Weiland, J.A. Southard, and S.L. Southard. 2005. Hydroacoustic Measurements During Pile Driving at the Hood Canal Bridge, September through November. 2004. Prepared for Washington State Department of Transportation by Battelle Marine Sciences Laboratory, Sequim, Washington.

Carrasquero, J. 2001. Over-Water Structures: Freshwater Issues. Prepared for Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation by Herrera Environmental Consultants, Seattle, Washington. April 2001.

Caudill, C.C., W.R. Daigle, M.L. Keefer, C.T. Boggs, M.A. Jepson, B.J. Burke, R.W. Zabel, T.C. Bjornn, and C.A. Peery. 2007. Slow Dam Passage in Adult Columbia River Salmonids Associated with Unsuccessful Migration: Delayed Effects of Passage Obstacles or Condition-Dependent Mortality? *Canadian Journal of Fisheries and Aquatic Sciences* 64(7): 979-995.

Cech, J.J., and C.E. Crocker. 2002. Physiology of Sturgeon: Effects of Hypoxia and Hypercapnia. *Journal of Applied Ichthyology* 18(4-6): 320-324.

Cederholm, C., R.E. Bilby, P. Bisson, T. Bumstead, B. Fransen, W. Scarlett, and J. Ward. 1997. Response of Juvenile Coho Salmon and Steelhead to Placement of Large Woody Debris in a Coastal Washington Stream. *North American Journal of Fisheries Management* 17: 947-963.

CEQ. 1997. Considering Cumulative Effects under the National Environmental Policy Act. Council on Environmental Quality.

Chambers, P.A., R.E. DeWreede, E.A. Irlandi, and H. Vandermeule. 1999. Management Issues in Aquatic Macrophyte Ecology: A Canadian Perspective. *Canadian Journal of Botany* 77(4): 471.

Chanseau, M., O. Croze, and M. Larinier. 1999. The Impact of Obstacles on the Pau River (France) on the Upstream Migration of Returning Adult Atlantic Salmon (*Salmo Salar L.*). *Bulletin Francais De La Peche Et De La Pisciculture* (353-54): 211-237.

Chapman, D.W. 1988. Critical Review of Variables Used to Define Effects of Fines in Redds of Large Salmonids. *Transactions of the American Fisheries Society* 117: 1-21.

Charland, J. 1998. Tide Gate Modifications for Fish Passage and Water Quality Enhancement. Garibaldi, Oregon: Tillamook Bay National Estuary Project.

Chen, J.Q., J.F. Franklin, and T.A. Spies. 1992. Vegetation Responses to Edge Environments in Old-Growth Douglas-Fir Forests. *Ecological Applications* 2(4): 387-396.

Chen, J.Q., J.F. Franklin, and T.A. Spies. 1993. Contrasting Microclimates among Clear-Cut, Edge, and Interior of Old-Growth Douglas-Fir Forest. *Agricultural and Forest Meteorology* 63(3-4): 219-237.

Chen, J.Q., J.F. Franklin, and T.A. Spies. 1995. Growing-Season Microclimatic Gradients from Clear-Cut Edges into Old-Growth Douglas-Fir Forests. *Ecological Applications* 5(1): 74-86.

Chen, J.Q., S.C. Saunders, T.R. Crow, R.J. Naiman, K.D. Brosofske, G.D. Mroz, B.L. Brookshire, and J.F. Franklin. 1999. Microclimate in Forest Ecosystem and Landscape Ecology - Variations in Local Climate Can Be Used to Monitor and Compare the Effects of Different Management Regimes. *Bioscience* 49(4): 288-297.

Cheng, W., and J.C. Chen. 2000. Effects of pH, Temperature and Salinity on Immune Parameters of the Freshwater Prawn *Macrobrachium Rosenbergii*. *Fish & Shellfish Immunology* 10(4): 387-391.

Chester, H., and R. Norris. 2006. Dams and Flow in the Cotter River, Australia: Effects on Instream Trophic Structure and Benthic Metabolism. *Hydrobiologia* 572: 275-286.

Chrzastowski, M.J., and T.A. Thompson. 1994. Late Wisconsinan and Holocene Geologic History of the Illinois-Indiana Coast of Lake-Michigan. *Journal of Great Lakes Research* 20(1): 9-26.

Church, M., M.A. Hassan, and J.F. Wolcott. 1998. Stabilizing Self-Organized Structures in Gravel-Bed Stream Channels: Field and Experimental Observations. *Water Resources Research* 34(11): 3169-3179.

Collier, M.P., R.H. Webb, and E.D. Andrews. 1997. Experimental Flooding in the Grand Canyon. *Scientific American* 276: 82-89.

Cook, M.A., K.M. Guthrie, M.B. Rust, and P.D. Plesha. 2005. Effects of Salinity and Temperature During Incubation on Hatching and Development of Lingcod *Ophiodon Elongatus* Girard, Embryos. *Aquaculture Research* 36(13): 1298-1303.

Cooper, A.C. 1965. The Effects of Transported Stream Sediments on the Survival of Sockeye and Pink Salmon Eggs and Alevins. International Pacific Salmon Fisheries Commission Bulletin 18. 71p.

Cooper, K., S. Boyd, J. Aldridge, and H. Rees. 2007. Cumulative Impacts of Aggregate Extraction on Seabed Macro-Invertebrate Communities in an Area Off the East Coast of the United Kingdom. *Journal of Sea Research* 57(4): 288-302.

Cordell, J.R. 1986. Structure and Dynamics of an Epibenthic Harpacticoid Assemblage and the Role of Predation by Juvenile Salmon. Seattle, Washington: University of Washington.

Cordova, J.M., E.J. Rosi-Marshall, A.M. Yamamuro, and G.A. Lamberti. 2007. Quantity, Controls and Functions of Large Woody Debris in Midwestern USA Streams. *River Research and Applications* 23(1): 21-33.

COSEWIC. 2003. Rocky Mountain Ridge Mussel, *Gonidea Angulata*. Status Report. Committee on the Status of Endangered Wildlife in Canada. Ottawa, Ontario: Canadian Wildlife Service, Environment Canada.

Couch, D., and T.J. Hassler. 1990. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest) Olympia Oyster. PBS Record: 115470. U.S. Army Corps of Engineers and U.S. Fish and Wildlife Service.

Counihan, T.D., A.I. Miller, M.G. Mesa, and M.J. Parsley. 1998. The Effects of Dissolved Gas Supersaturation on White Sturgeon Larvae. *Transactions of the American Fisheries Society* 127(2): 316-322.

Coutant, C.C. 2004. A Riparian Habitat Hypothesis for Successful Reproduction of White Sturgeon. *Reviews in Fisheries Science* 12(1): 23-73.

Cox, M., P.H. Rogers, A.N. Popper, and W.M. Saidel. 1987. Anatomical Effects of Intense Tone Stimulation in the Goldfish Ear: Dependence on Sound-Pressure Level and Frequency. *Journal of the Acoustical Society of America.*

Crocker, C.E., and J.J. Cech. 1997. Effects of Environmental Hypoxia on Oxygen Consumption Rate and Swimming Activity in Juvenile White Sturgeon, *Acipenser Transmontanus*, in Relation to Temperature and Life Intervals. *Environmental Biology of Fishes* 50(4): 383-389.

Cushing, C.E., and J.D. Allan. 2001. *Streams: Their Ecology and Life*. San Diego, California: Academic Press.

Dadswell, M.J. 1996. The Removal of Edwards Dam, Kennebec River, Maine: Its Effects on the Restoration of Anadromous Fishes. Draft environmental impact statement.

Dalbey, S.R., T.E. McMahon, and W. Fredenberg. 1996. Effect of Electrofishing Pulse Shape and Electrofishing ‑ Induced Spinal Injury to Long-Term Growth and Survival of Wild Rainbow Trout. *North American Journal of Fisheries Management* 16: 560-569.

de Croux, P., M. Julieta, and A. Loteste. 2004. Lethal Effects of Elevated pH and Ammonia on Juveniles of Neotropical Fish *Colosoma Macropomum* (Pisces, Caracidae). *Journal of Environmental Biology* 25(1): 7-10.

deBruyn, A.M.H., D.J. Marcogliese, and J.B. Rasmussen. 2003. The Role of Sewage in a Large River Food Web. *Canadian Journal of Fisheries and Aquatic Sciences* 60(11): 1332-1344.

Dernie, K.M., M.J. Kaiser, E.A. Richardson, and R.M. Warwick. 2002. Recovery of Soft Sediment Communities and Habitats Following Physical Disturbance. *Journal of Experimental Marine Biology and Ecology* 4069: 1-20.

Desbonnet, A., L.P. Pogue, D.Resiss, J. Boyd, J. Williams, and M. Imperial. 1995. Development of Coastal Vegetated Buffer Programs. *Coastal Management* 23: 91-109.

Desbonnet, A., P. Pogue, V. Lee, and N. Wolff. 1994. Vegetated Buffers in the Coastal Zone - a Summary Review and Bibliography. University of Rhode Island Graduate School of Oceanography.

Dewson, Z.S., A.B.W. James, and R.G. Death. 2007. Invertebrate Responses to Short-Term Water Abstraction in Small New Zealand Streams. *Freshwater Biology* 52(2): 357-369.

DFO. 2007. Concrete Wash Water: Characteristics. Canada Department of Fisheries and Oceans. Available at: http://www-heb.pac.dfo-mpo.gc.ca/water_quality/fish_and_pollution/conc_char_e.htm (accessed June 3, 2007).

Dietrich, W.E., J.W. Kirchner, H. Ikeda, and F. Iseya. 1989. Sediment Supply and Development of Coarse Surface Layer in Gravel Bedded Rivers. *Nature* 340(6230): 215-217.

Diez, J.R., S. Larranaga, A. Elosegi, and J. Pozo. 2000. Effect of Removal of Wood on Streambed Stability and Retention of Organic Matter. *Journal of the North American Benthological Society* 19(4): 621-632.

Dill, L.M., R.C. Ydenberg, and A.H.G. Fraser. 1981. Food Abundance and Territory Size in Juvenile Coho Salmon (*Oncorhynchus Kisutch*). *Canadian Journal of Zoology* 59: 1801-1809.

Doeg, T.J., and J.D. Koehn. 1994. Effects of Draining and Desilting a Small Weir on Downstream Fish and Macroinvertebrates. *Regulated Rivers-Research & Management* 9(4): 263-277.

DOI. 1995. Final Environmental Impact Statement: Elwha River Ecosystem Restoration, Olympic National Park, Washington. Department of the Interior.

Dooley, K.M., C.F. Knopf, and R.P. Gambrell. 1999. Final Report: pH-Neutral Concrete for Attached Microalgae and Enhanced Carbon Dioxide Fixation - Phase I. Baton Rouge, Louisiana: Louisiana State University.

Downing, J. 1983. *The Coast of Puget Sound: Its Processes and Development*. Seattle, Washington: Washington SeaGrant Program. University of Washington Press.

Downing, J.A., Y. Rochon, M. Perusse, and H. Harvey. 1993. Spatial Aggregation, Body Size, and Reproductive Success in the Freshwater Mussel *Elliptio Complanata*. *Journal of the North American Benthological Society* 12: 148-156.

Drinkwater, K.F., and K.T. Frank. 1994. Effects of River Regulation and Diversion on Marine Fish and Invertebrates. *Aquatic Conservation-Marine and Freshwater Ecosystems* 4(2): 135-151.

Dudley, S.J., J.C. Fischenich, and S.R. Abt. 1998. Effect of Woody Debris Entrapment on Flow Resistance. *Journal of the American Water Resources Association* 34(5): 1189-1197.

Dunn, J.R., and A.C. Matarese. 1987. A Review of Early Life History of Northeast Pacific Gadoid Fishes. *Fisheries Research* 5: 163-184.

Dwyer, W.P., and R.G. White. 1997. Effect of Electroshock on Juvenile Arctic Grayling and Yellowstone Cutthroat Trout Growth 100 Days after Treatment. *North American Journal of Fisheries Management* 17: 174-177.

Eby, L.A., L.B. Crowder, C.M. McClellan, C.H. Peterson, and M.J. Powers. 2005. Habitat Degradation from Intermittent Hypoxia: Impacts on Demersal Fishes. *Marine Ecology-Progress Series* 291: 249-261.

Ecology. 1999. Working in the Water. Washington State Department of Ecology.

Ecology. 2002. Evaluating Criteria for the Protection of Freshwater Aquatic Life in Washington's Surface Water Quality Standards, Dissolved Oxygen: Draft Discussion Paper and Literature Summary. Publication Number 00-10-071. Olympia, Washington: Washington State Department of Ecology.

Edinger, J.E., and V. S. Kolluru. 2000. Power Plant Intake Entrainment Analysis. *Journal of Energy Engineering-ASCE* 126 (1): 1-14.

Edwards, R.T. 1998. The Hyporheic Zone. In *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*, edited by R.J. Naiman and R.E. Bilby. New York: Springer-Verlag.

Eilers, H.P.I. 1975. Plants, Plant Communities, Net Production and Tide Levels: The Ecological Biogeography of the Nehalem Salt Marshes, Tillamook County, Oregon. Oregon State University, Corvallis, Oregon, 368 pp.

Ellis, M.M. 1936. Erosion Silt as a Factor in Aquatic Environments. *Ecology* 17(1): 29-42.

Embrey, S.S., and P.W. Moran. 2006. Quality of Streamwater in the Puget Sound Basin–a Decade of Study and Beyond [Poster]. *Toxics in Puget Sound: Connecting Marine Environment to Human Health and the Economy, Puget Sound Action Team Forum*, Seattle, Washington, April 5, 2006.

Emmett, R.L., S.L. Stone, S.A. Hinton, and M.E. Monaco. 1991. Distribution and Abundance of Fishes and Invertebrates in West Coast Estuaries, Volume II: Species Life History Summaries. Rockville, Maryland: NOAA/NOA Strategic Environmental Assessments Division.

Enger, P.S. 1981. Frequency Discrimination in Teleosts - Central or Peripheral? In *Hearing and Sound Communication in Fishes*, edited by W.N. Tavolga, A.N. Popper and R.R. Fay. New York: Springer-Verlag. pp. 243-255.

Ensign, S.H., and M.W. Doyle. 2005. In-Channel Transient Storage and Associated Nutrient Retention: Evidence from Experimental Manipulations. *Limnology and Oceanography* 50(6): 1740-1751.

Erftemeijer, P.L.A., and R.R.R. Lewis. 2006. Environmental Impacts of Dredging on Seagrasses: A Review. *Marine Pollution Bulletin* 52(12): 1553-1572.

Erickson, D.L., J.A. North, J.E. Hightower, J. Weber, and L. Lauck. 2002. Movement and Habitat Use of Green Sturgeon *Acipenser Medirostris* in the Rogue River, Oregon, USA. *Journal of Applied Ichthyology* 18(4-6): 565-569.

Eschmeyer, W.N., E.S. Herald, and H. Hammon. 1983. *A Field Guide to Pacific Coast Fishes of North America*. Boston, Massachusetts: Houghton Mifflin.

Everett, R.A., and G.M. Ruiz. 1993. Coarse Woody Debris as a Refuge from Predation in Aquatic Communities. *Oecologia* 93(4): 475-486.

Fausch, K.D., C. Gowan, A.D. Richmond, and S.C. Riley. 1995. The Role of Dispersal in Trout Population Response to Habitat Formed by Large Woody Debris in Colorado Mountain Streams. *Bulletin Francais De La Peche Et De La Pisciculture* (337-9): 179-190.

Faustini, J.M., and J.A. Jones. 2003. Influence of Large Woody Debris on Channel Morphology and Dynamics in Steep, Boulder-Rich Mountain Streams, Western Cascades, Oregon. *Geomorphology* 51(1-3): 187-205.

Fay, R.R. 1988. Peripheral Adaptations for Spatial Hearing in Fish. In *Sensory Biology of Aquatic Animals*. New York: Springer-Verlag. pp. 711-731.

Feist, B.E., J. Anderson, and R. Miyamoto. 1992. Potential Impacts of Pile Driving on Juvenile Pink (*Oncorhynchus Gorbuscha*) and Chum (*O. Keta*) Salmon Behavior and Distribution. Seattle, Washington: University of Washington.

Feist, G.W., M.A.H. Webb, D.T. Gundersen, E.P. Foster, C.B. Schreck, A.G. Maule, and M.S. Fitzpatrick. 2005. Evidence of Detrimental Effects of Environmental Contaminants on Growth and Reproductive Physiology of White Sturgeon in Impounded Areas of the Columbia River. *Environmental Health Perspectives* 113(12): 1675-1682.

Feller, M.C. 2005. Forest Harvesting and Streamwater Inorganic Chemistry in Western North America: A Review. *Journal of the American Water Resources Association* 41(4): 785-811.

Fernald, A.G., D.H. Landers, and P.J. Wigington. 2006. Water Quality Changes in Hyporheic Flow Paths between a Large Gravel Bed River and Off-Channel Alcoves in Oregon, USA. *River Research and Applications* 22(10): 1111-1124.

Ferraro, S.P., and F.A. Cole. 2007. Benthic Macrofauna-Habitat Associations in Willapa Bay, Washington, USA. *Estuarine Coastal and Shelf Science* 71(3-4): 491-507.

Feyrer, F., T. Sommer, and W. Harrell. 2006. Importance of Flood Dynamics Versus Intrinsic Physical Habitat in Structuring Fish Communities: Evidence from Two Adjacent Engineered Floodplains on the Sacramento River, California. *North American Journal of Fisheries Management* 26(2): 408-417.

Finlayson, D. 2006. The Geomorphology of Puget Sound Beaches. University of Washington, Seattle, 216 pp.

Fischer, H.B., List, E. J., Koh, R. C. Y., Imberger, J., Brooks, N. H. 1979. *Mixing in Inland and Coastal Waters*. San Diego, California: Academic Press.

Flett, P.A., K.R. Munkittrick, G. VanDerKraak, and J.F. Leatherland. 1996. Overripening as the Cause of Low Survival to Hatch in Lake Erie Coho Salmon (*Oncorhynchus Kisutch*) Embryos. *Canadian Journal of Zoology-Revue Canadienne De Zoologie* 74(5): 851-857.

Florsheim, J.L., and J.F. Mount. 2002. Restoration of Floodplain Topography by Sand-Splay Complex Formation in Response to Intentional Levee Breaches, Lower Cosumnes River, California. *Geomorphology* 44: 67-94.

Fonseca, M.S., and S.S. Bell. 1998. Influence of Physical Setting on Seagrass Landscapes near Beaufort, North Carolina, USA. *Marine Ecology-Progress Series* 171: 109-121.

Fonseca, M.S., J.C. Zieman, G.W. Thayer, and J.S. Fisher. 1983. The Role of Current Velocity in Structuring Eelgrass (*Zostera-Marina L*) Meadows. *Estuarine Coastal and Shelf Science* 17(4): 367-380.

Fredenberg, W.A. 1992. Evaluation of Electrofishing-Induced Spinal Injuries Resulting from Field Electrofishing Surveys in Montana. Montana Department of Fish, Wildlife and Parks, Helena, Montana.

Frest, T.J., and E.J. Johannes. 1995. Freshwater Molluscs of the Upper Sacramento System, California with Particular Reference to the Cantara Spill. Sacramento, California: State of California, Department of Fish and Game.

Fries, L.T., and D.E. Bowles. 2002. Water Quality and Macroinvertebrate Community Structure Associated with a Sportfish Hatchery Outfall. *North American Journal of Aquaculture* 64(4): 257-266.

Frisch, A.J., and T.A. Anderson. 2000. The Response of Coral Trout (*Plectropomus Leopardus*) to Capture, Handling and Transport and Shallow Water Stress. *Fish Physiology and Biochemistry* 23(1): 23-24.

Fritioff, A., and M. Greger. 2003. Aquatic and Terrestrial Plant Species with Potential to Remove Heavy Metals from Stormwater. *International Journal of Phytoremediation* 5(3): 211-224.

Fuchs, S.A., S.G. Hinch, and E. Mellina. 2003. Effects of Streamside Logging on Stream Macroinvertebrate Communities and Habitat in the Sub-Boreal Forests of British Columbia, Canada. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere* 33(8): 1408-1415.

Gadomski, D.M., and M.J. Parsley. 2005. Effects of Turbidity, Light Level, and Cover on Predation of White Sturgeon Larvae by Prickly Sculpins. *Transactions of the American Fisheries Society* 134(2): 369-374.

Gagnaire, B., H. Frouin, K. Moreau, H. Thomas-Guyon, and T. Renault. 2006. Effects of Temperature and Salinity on Haemocyte Activities of the Pacific Oyster, *Crassostrea Gigas* (Thunberg). *Fish & Shellfish Immunology* 20(4): 536-547.

Gallardo, A.H., and A. Marui. 2006. Submarine groundwater discharge: an outlook of recent advances and current knowledge. *Geo-Marine Letters*. Vol. 26. No. 2 June 2006: 102-113.

Gandy, C.J., J.W.N. Smith, and A.P. Jarvis. 2007. Attenuation of Mining-Derived Pollutants in the Hyporheic Zone: A Review. *Science of the Total Environment* 373(2-3): 435-446.

Gardner, F. 1981. Washington Coastal Areas of Major Biological Significance. Olympia, Washington: Washington State Department of Ecology, Baseline Studies Program.

Garland, R.D., K.F. Tiffan, D.W. Rondorf, and L.O. Clark. 2002. Comparison of Subyearling Fall Chinook Salmon's Use of Riprap, Revetments and Unaltered Habitats in Lake Wallula of the Columbia River. *North American Journal of Fisheries Management* 22(4): 1283-1289.

Garrison, K.J., and B.S. Miller. 1982. Review of the Early Life History of Puget Sound Fishes. Seattle, Washington: University of Washington, Fish. Res. Inst.

Gay, P., G. Vellidis, and J.J. Delfino. 2006. The Attenuation of Atrazine and Its Major Degradation Products in a Restored Riparian Buffer. *Transactions of the ASABE* 49(5): 1323-1339.

Geist, D.R. 2000. Hyporheic Discharge of River Water into Fall Chinook Salmon (*Oncorhynchus Tshawytscha*) Spawning Areas in the Hanford Reach, Columbia River. *Canadian Journal of Fisheries and Aquatic Science* 57: 1647-1656.

Geist, D.R., R.S. Brown, V. Cullinan, S.R. Brink, K. Lepla, P. Bates, and J.A. Chandler. 2005. Movement, Swimming Speed, and Oxygen Consumption of Juvenile White Sturgeon in Response to Changing Flow, Water Temperature, and Light Level in the Snake River, Idaho. *Transactions of the American Fisheries Society* 134(4): 803-816.

Giannico, G., and J.A. Souder. 2005. Tide Gates in the Pacific Northwest: Operation, Types and Environmental Effects. ORESU-T-05-001. Corvallis, Oregon: Oregon Sea Grant.

Gillilan, D.M., and T.C. Brown. 1997. *Instream Flow Protection: Seeking a Balance in Western Water Use*. Washington, DC: Island Press.

Giorgi, A.E. 1981. The Environmental Biology of the Embryos, Egg Masses and Nesting Sites of the Lingcod, *Ophiodon Elongatus*. Seattle, Washington: National Marine Fisheries Service, NWAFC Proc. Rept.

Goetz, F.A., E. Jeanes, E. Beamer, G. Hart, C. Morello, M. Camby, C. Ebel, E. Conner, and H. Berge. 2004. Bull Trout in the Nearshore (Preliminary Draft). U.S. Army Corps of Engineers, Seattle District.

Gonor, J., J. Sedell, and P. Benner. 1988. What We Know About Large Trees in Estuaries, in the Sea, and on Coastal Beaches. In *From the Forest to the Sea: A Story of Fallen Trees*, edited by C. Maser, R. Tarrant, J. Trappe and J. Franklin. Portland, Oregon. USDA Forest Service, Pacific Northwest Research Station. pp. 83-112.

Gosset, C., J. Rives, and J. Labonne. 2006. Effect of Habitat Fragmentation on Spawning Migration of Brown Trout (*Salmo Trutta L.*). *Ecology of Freshwater Fish* 15(3): 247-254.

Gotthardt, T. 2006. Longfin Smelt. Anchorage, Alaska: Alaska Natural Heritage Program.

Goudreau, S.E., R.J. Neves, and R.J. Sheehan. 1993. Effects of Waste-Water Treatment-Plant Effluents on Fresh-Water Mollusks in the Upper Clinch River, Virginia, USA. *Hydrobiologia* 252(3): 211-230.

Grant, J., and B. Thorpe. 1991. Effects of Suspended Sediment on Growth, Respiration, and Excretion of the Soft-Shell Clam (*Mya-Arenaria*). *Canadian Journal of Fisheries and Aquatic Sciences* 48(7): 1285-1292.

Grant, J., C.T. Enright, and A. Griswold. 1990. Resuspension and Growth of *Ostrea-Edulis* - a Field Experiment. *Marine Biology* 104(1): 51-59.

Grapentine, L., Q. Rochfort, and J. Marsalek. 2004. Benthic Responses to Wet-Weather Discharges in Urban Streams in Southern Ontario. *Water Quality Research Journal of Canada* 39(4): 374-391.

Gray, J.S., R.S.S. Wu, and Y.Y. Or. 2002. Effects of Hypoxia and Organic Enrichment on the Coastal Marine Environment. *Marine Ecology-Progress Series* 238: 249-279.

Greathouse, E.A., C.M. Pringle, W.H. McDowell, and J.G. Holmquist. 2006. Indirect Upstream Effects of Dams: Consequences of Migratory Consumer Extirpation in Puerto Rico. *Ecological Applications* 16(1): 339-352.

Gregory, R.S. 1993. Effect of Turbidity on the Predator Avoidance-Behavior of Juvenile Chinook Salmon (*Oncorhynchus-Tshawytscha*). *Canadian Journal of Fisheries and Aquatic Sciences* 50(2): 241-246.

Gregory, R.S., and C.D. Levings. 1998. Turbidity Reduces Predation on Migrating Juvenile Pacific Salmon. *Transactions of the American Fisheries Society* 127(2): 275-285.

Greig, S.M., D.A. Sear, and P.A. Carling. 2007. A Review of Factors Influencing the Availability of Dissolved Oxygen to Incubating Salmonid Embryos. *Hydrological Processes* 21(3): 323-334.

Griffin, F.J., M.C. Pillai, C.A. Vines, J. Kaaria, T. Hibbard-Robbins, R. Yanagimachi, and G.N. Cherr. 1998. Effects of Salinity on Sperm Motility, Fertilization, and Development in the Pacific Herring, *Clupea Pallasi*. *Biological Bulletin* 194(1): 25-35.

Groberg, W.J., R.H. Mccoy, K.S. Pilcher, and J.L. Fryer. 1978. Relation of Water Temperature to Infections of Coho Salmon (*Oncorhynchus-Kisutch*), Chinook Salmon (*O-Tshawytscha*), and Steelhead Trout (*Salmo-Gairdneri*) with *Aeromonas-Salmonicida* and *Aeromanas-Hydrophila*. *Journal of the Fisheries Research Board of Canada* 35(1): 1-7.

Groot, C., and L. Margolis. 1991. *Pacific Salmon Life Histories*. Vancouver, British Columbia: University of British Columbia Press.

Guerra-García, J.M., J. Corzo, and J.C. García-Gómez. 2003. Short-Term Benthic Recolonization after Dredging in the Harbour of Ceuta, North Africa. *Marine Ecology* 24(3): 217-229.

Guidetti, P., L. Verginella, C. Viva, R. Odorico, and F. Boero. 2005. Protection Effects on Fish Assemblages, and Comparison of Two Visual-Census Techniques in Shallow Artificial Rocky Habitats in the Northern Adriatic Sea. *Journal of the Marine Biological Association of the United Kingdom* 85(2): 247-255.

Guidetti, P., S. Bussotti, and F. Boero. 2005. Evaluating the Effects of Protection on Fish Predators and Sea Urchins in Shallow Artificial Rocky Habitats: A Case Study in the Northern Adriatic Sea. *Marine Environmental Research* 59(4): 333-348.

Gurnell, A., and G. Petts. 2006. Trees as Riparian Engineers: The Tagliamento River, Italy. *Earth Surface Processes and Landforms* 31(12): 1558-1574.

Gustafson, R.G., T.C. Wainwright, G.A. Winans, F.W. Waknitz, L.T. Parker, and R.S. Waples. 1997. Status Review of Sockeye Salmon from Washington and Oregon, NOAA Technical Memorandum. Seattle, Washington: Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Haas, M.E., C.A. Simenstad, J.R. Cordell, D.A. Beauchamp, and B.S. Miller. 2002. Effects of Large Overwater Structures on Epibenthic Juvenile Salmon Prey Assemblages in Puget Sound, Washington. Washington State Transportation Center (TRAC).

Haldorson, L., and L.J. Richards. 1986. Post-Larval Copper Rockfish in the Strait of Georgia: Habitat Use, Feeding, and Growth in the First Year. In *Proceedings International Rockfish Symposium,* Anchorage, Alaska: Alaska Sea Grant College Program. Cited in NRC 2001. pp. 129-141.

Hall, M.O., M.J. Durako, J.W. Fourqurean, and J.C. Zieman. 1999. Decadal Changes in Seagrass Distribution and Abundance in Florida Bay. *Estuaries* 22(2B): 445-459.

Hallock, M., and P.E. Mongillo. 1998. Washington State Status Report for the Pygmy Whitefish. Olympia, Washington: Washington Department of Fish and Wildlife, Fish Management Program, Freshwater Resource Division.

Hansen, J.A., P.G. Welsh, J. Lipton, and D. Cacela. 2002. Effects of Copper Exposure on Growth and Survival of Juvenile Bull Trout. *Transactions of the American Fisheries Society* 131(4): 690-697.

Hansen, J.A., P.G. Welsh, J. Lipton, and M.J. Suedkamp. 2002. The Effects of Long-Term Cadmium Exposure on the Growth and Survival of Juvenile Bull Trout (*Salvelinus Confluentus*). *Aquatic Toxicology* 58(3-4): 165-174.

Hansen, J.A., P.G. Welsh, J. Lipton, D. Cacela, and A.D. Dailey. 2002. Relative Sensitivity of Bull Trout (*Salvelinus Confluentus*) and Rainbow Trout (*Oncorhynchus Mykiss*) to Acute Exposures of Cadmium and Zinc. *Environmental Toxicology and Chemistry* 21(1): 67-75.

Hard, J.J., R.G. Kope, W.S. Grant, F.W. Waknitz, L.T. Parker, and R.S. Waples. 1996. Status Review of Pink Salmon from Washington, Oregon, California. Seattle, Washington: Northwest Fisheries Science Center.

Hardyniec, S., and S. Skeen. 2005. Pile Driving and Barotraumas Effects. No. 1941. *Journal of Transportation Research Board*.

Harrahy, L.N.M., C.B. Schreck, and A.G. Maule. 2001. Antibody-Producing Cells Correlated to Body Weight in Juvenile Chinook Salmon (*Oncorhynchus Tshawytscha*) Acclimated to Optimal and Elevated Temperatures. *Fish & Shellfish Immunology* 11(8): 653-659.

Harris, C. 1974. The Geographical Distribution and Habitat of the Olympic Mudminnow (*Novumbra Hubbsi*). Master's Thesis, University of Washington, Seattle, Washington.

Hart, J.L. 1973. Pacific Fishes of Canada. *Fisheries Research Board of Canada* 180: 730.

Harvey, B.C., R.J. Nakamoto, and J.L. White. 1999. Influence of Large Woody Debris and a Bankfull Flood on Movement of Adult Resident Coastal Cutthroat Trout (*Oncorhynchus Clarki*) During Fall and winter. *Canadian Journal of Fisheries and Aquatic Sciences* 56(11): 2161-2166.

Hastings, M.C. 1995. Physical Effects of Noise on Fishes. *Proceedings of INTER-NOISE 95, The 1995 International Congress on Noise Control Engineering*, pp. 979–984.

Hastings, M.C., A.N. Popper, J.J. Finneran, and P.J. Lanford. 1996. Effects of Low-Frequency Underwater Sound on Hair Cells of the Inner Ear and Lateral Line of the Teleost Fish *Astronotus Ocellatus*. *Journal of the Acoustical Society of America* 99(3): 1759-1766.

Hastings, M.C., and A.N. Popper. 2005. Effects of Sound of Fish. Prepared for California Department of Transportation by Jones and Stokes, Sacramento, California. August 2005.

Hatch, A.C., and G.A.J. Burton. 1999. Sediment Toxicity and Stormwater Runoff in a Contaminated Receiving System: Consideration of Different Bioassays in the Laboratory and Field. *Chemospher* 39: 1001-1017.

Hauer, F.R., G.C. Poole, J.T. Gangemi, and C.V. Baxter. 1999. Large Woody Debris in Bull Trout (*Salvelinus Confluentus*) Spawning Streams of Logged and Wilderness Watersheds in Northwest Montana. *Canadian Journal of Fisheries and Aquatic Sciences* 56(6): 915-924.

Hawkins, A.D. 1986. Underwater Sound and Fish Behavior. In *The Behavior of Teleost Fishes*, edited by T.J. Pitcher. Maryland: Johns Hopkins University Press. pp. 114-151.

Hawkins, A.D., and A.D.F. Johnstone. 1978. The Hearing of the Atlantic Salmon, *Salmo Salar*. *Journal of Fish Biology* 13: 655-673.

Hawkins, C.P., and J.R. Sedell. 1981. Longitudinal and Seasonal-Changes in Functional-Organization of Macroinvertebrate Communities in 4 Oregon Streams. *Ecology* 62(2): 387-397.

Hawkins, C.P., M.L. Murphy, N.H. Anderson, and M.A. Wilzbach. 1983. Density of Fish and Salamanders in Relation to Riparian Canopy and Physical Habitat in Streams of the Northwestern United-States. *Canadian Journal of Fisheries and Aquatic Sciences* 40(8): 1173-1185.

Hayman, R.A., E.M. Beamer, and R.E. McClure. 1996. Skagit System Cooperative Chinook Restoration Research Progress Report No. 1. FY 1995 Skagit River Chinook Restoration Research. LaConner, Washington: Skagit System Cooperative.

Hazelwood, R.A., and J. Connelly. 2005. Estimation of Underwater Noise - a Simplified Method. *International Journal of the Society of Underwater Technology* 26(3): 51-57.

Healey, M.C. 1982. Juvenile Pacific Salmon in Estuaries: The Life Support System. In *Estuarine Comparisons*, edited by V.S. Kennedy. New York, New York: Academic Press. pp. 315-341.

Healey, M.C. 1991. Life History of Chinook Salmon (*Oncorhynchus Tshawytscha*). In *Pacific Salmon Life Histories*, edited by C. Groot and L. Margolis. Vancouver, British Columbia: University of British Columbia Press. pp. 311-394230.

Heard, W.R. 1991. Life History of Pink Salmon (*Oncorhynchus Gorbuscha*). In *Pacific Salmon Life Histories*, edited by C. Groot and L. Margolis. Vancouver, British Columbia: University of British Columbia Press. pp. 120-230.

Heath, A.G., and G.M. Hughes. 1973. Cardiovascular and Respiratory Changes During Heat Stress in Rainbow-Trout (*Salmo-Gairdneri*). *Journal of Experimental Biology* 59(2): 323-338.

Heathershaw, A.D., P.D. Ward, and A.M. David. 2001. The Environmental Impact of Underwater Sound *Proc. I.O.A.* 23: 1-13.

Hecht, S.A., D.H. Baldwin, C.A. Mebane, T. Hawkes, and S.J. Gross. 2007. An Overview of Sensory Effects on Juvenile Salmonids Exposed to Dissolved Copper: Applying a Benchmark Concentration Approach to Evaluate Sublethal Neurobehavioral Toxicity. Technical White Paper. Lacey, Washington: National Marine Fisheries Service.

Heggenes, J., and R. Borgstrom. 1988. Effect of Mink, *Mustela Vison* Schreber, Predation on Cohorts of Juvenile Atlantic Salmon, *Salmo Salar L.*, and Brown Trout, *S. Trutta L.*, in Three Small Streams. *Journal of Fish Biology* 33(6): 885-894.

Henning, J. 2004. An Evaluation of Fish and Amphibian Use of Restored and Natural Floodplain Wetlands. Olympia, Washington: Washington Department of Fish and Wildlife.

Hernandez-Miranda, E., A.T. Palma, and F.P. Ojeda. 2003. Larval Fish Assemblages in Nearshore Coastal Waters Off Central Chile: Temporal and Spatial Patterns. *Estuarine Coastal and Shelf Science* 56(5-6): 1075-1092.

Herrera. 2005. Marine Shoreline Sediment Survey and Assessment, Thurston County, Washington. Prepared for Thurston Regional Planning Council by Herrera Environmental Consultants, Inc., Seattle, Washington.

Herrera. 2006a. Landslide Disposal Site Identification: Sound Transit Everett-to-Seattle Commuter Rail. Seattle, Washington: Sound Transit.

Herrera. 2006b. Simonson Place Boat Ramp and Stormwater Outfall Replacement Project: Coastal Processes Assessment. Seattle, Washington: Island County Public Works.

Herrera. 2007a. Fish Passage White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, November 2007.

Herrera. 2007b. Fish Screens White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, November 2007.

Herrera. 2007c. Marinas White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, June 2007.

Herrera. 2007d. Water Quality Statistical and Pollutant Loading Analysis: Green-Duwamish Watershed Water Quality Assessment. Prepared for King County Department of Natural Resources and Parks by Herrera Environmental Consultants, Inc. (Herrera), Seattle, Washington.

Herrera. 2007e. Habitat Modifications White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, August 2007.

Herrera. 2007f. Shoreline Modifications White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, July 2007.

Hershey, A.E., and G.A. Lamberti. 1992. Stream Macroinvertebrate Communities. In *Watershed Management – Balancing Sustainability and Environmental Change*, edited by R.J. Naiman. New York: Springer-Verlag.

Hetrick, N.J., M.A. Brusven, T.C. Bjornn, R.M. Keith, and W.R. Meehan. 1998. Effects of Canopy Removal on Invertebrates and Diet of Juvenile Coho Salmon in a Small Stream in Southeast Alaska. *Transactions of the American Fisheries Society* 127(6): 876-888.

Hewitt, M., R. Schryer, A. Pryce, A. Belknap, B. Firth, and G. Van Der Kraak. 2005. Accumulation of Hormonally Active Substances by Wild White Sucker (*Catostomus Commersoni*) Exposed to Effluents Discharged to the Wabigoon River. *Water Quality Research Journal of Canada* 40(3): 315-327.

Hicks, B.J., J.D. Hall, P.A. Bisson, and J.R. Sedell. 1991. Responses of Salmonids to Habitat Change. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*, edited by W.R. Meehan. Bethesda, Maryland: American Fisheries Society Special Publication 19.

Higgins, K., P. Schlenger, J. Small, D. Hennessy, and J. Hall. 2005. Spatial Relationships between Beneficial and Detrimental Nearshore Habitat Parameters in WRIA 9 and the City of Seattle. *Proceedings of the 2005 Puget Sound Georgia Basin Research Conference.*

Hilderbrand, R.H., A.D. Lemly, C.A. Dolloff, and K.L. Harpster. 1997. Effects of Large Woody Debris Placement on Stream Channels and Benthic Macroinvertebrates. *Canadian Journal of Fisheries and Aquatic Sciences* 54(4): 931-939.

Hill, M.J., E.A. Long, and S. Hardin. 1993. Effects of Dam Removal on Dead Lake, Chipola River, Florida. Apalachicola River Watershed Investigations F-39-R. Florida Game and Fresh Water Fish Commission.

Hinchey, E.K., L.C. Schaffner, C.C. Hoar, B.W. Vogt, and L.P. Batte. 2006. Responses of Estuarine Benthic Invertebrates to Sediment Burial: The Importance of Mobility and Adaptation. *Hydrobiologia* 556: 85-98.

Hinton, S.A., R.L. Emmett, and G.T. McCabe. 1992. Benthic Invertebrates, Demersal Fishes and Sediment Characteristics at and Adjacent to Ocean Dredge Material Disposal Site F, Offshore from the Columbia River, June 1989-1990.

Hirose, T., and K. Kawaguchi. 1998. Sediment Size Composition as an Important Factor in the Selection of Spawning Site by the Japanese Surf Smelt *Hypomesus Japonicus*. *Fisheries Science* 64(6): 995-996.

Hoffmann, A. 2000. The Association of the Stream Caddisfly *Lasiocephala Basalis* (Kol.) (Trichoptera: Lepidostomatidae) with Wood. *International Review of Hydrobiology* 85(1): 79-93.

Holliman, F.M., J.B. Reynolds, and T.J. Kwak. 2003. Electroshock-Induced Injury and Mortality in the Spotfin Chub, a Threatened Minnow. *North American Journal of Fish Management* 23(3): 962-966.

Holm, T.E., and P. Clausen. 2006. Effects of Water Level Management on Autumn Staging Waterbird and Macrophyte Diversity in Three Danish Coastal Lagoons. *Biodiversity and Conservation* 15(14): 4399-4423.

Hood, W.G. 2004. Indirect Environmental Effects of Dikes on Estuarine Tidal Channels: Thinking Outside of the Dike for Habitat Restoration and Monitoring. *Estuaries* 27(2): 273-282.

Hood, W.G. 2006. A Conceptual Model of Depositional, Rather Than Erosional, Tidal Channel Development in the Rapidly Prograding Skagit River Delta (Washington, USA). *Earth Surface Processes and Landforms* 31(14): 1824-1838.

Horner, R.A. 1998. Harmful Algal Blooms in Puget Sound: General Perspective. *Puget Sound Research: From Basic Science to Resource Management*, Seattle, Washington, March 12 and 13, 1998. pp. 809-811.

Hoshikawa, H., Y. Sakai, and A. Kijima. 1998. Growth Characteristics of the Hybrid between Pinto Abalone, *Haliotis Kamtschatkana* Jonas, and Exo Abalone, *H-Discus* Hannai Ino, under High and Low Temperature. *Journal of Shellfish Research* 17(3): 673-677.

Howard, J.K., and K.M. Cuffey. 2003. Freshwater Mussels in a California North Coast Range River: Occurrence, Distribution, and Controls. *Journal of the North American Benthological Society* 22(1): 63-77.

Howell, M.D., M.D. Romano, and T.A. Rien. 2001. Draft Outmigration Timing and Distribution of Larval Eulachon, *Thaleichthys Pacificus*, in the Lower Columbia River, Spring 2001. Washington Department of Fish and Wildlife and Oregon Department of Fish and Wildlife.

Hughes, G.W., and A.E. Peden. 1989. Status of the Umatilla Dace, *Rhinichthys-Umatilla*, in Canada. *Canadian Field-Naturalist* 103(2): 193-200.

Illingworth and Rodkin, Inc. 2001. Noise and Vibration Measurements Associated with the Pile Installation Demonstration Project for the San Francisco-Oakland Bay Bridge East Span, Final Data Report.

Ingermann, R.L., M. Holcomb, M.L. Robinson, and J.G. Cloud. 2002. Carbon Dioxide and pH Affect Sperm Motility of White Sturgeon (*Acipenser Transmontanus*). *Journal of Experimental Biology* 205(18): 2885-2890.

Jackson, C.R., C.A. Sturm, and J.M. Ward. 2001. Timber Harvest Impacts on Small Headwater Stream Channels in the Coast Ranges of Washington. *Journal of the American Water Resources Association* 37(6): 1533-1549.

Jager, H.I. 2006. Chutes and Ladders and Other Games We Play with Rivers. I. Simulated Effects of Upstream Passage on White Sturgeon. *Canadian Journal of Fisheries and Aquatic Sciences* 63(1): 165-175.

Johnson, G.E., B.D. Ebberts, D.D. Dauble, A.E. Giorgi, P.G. Heisey, R.P. Mueller, and D.A. Neitzel. 2003. Effects of Jet Entry at High-Flow Outfalls on Juvenile Pacific Salmon. *North American Journal of Fisheries Management* 23(2): 441-449.

Johnson, L.L., G.M. Ylitalo, M.R. Arkoosh, A.N. Kagley, C. Stafford, J.L. Bolton, J. Buzitis, B.F. Anulacion, and T.K. Collier. 2007. Contaminant Exposure in Outmigrant Juvenile Salmon from Pacific Northwest Estuaries of the United States. *Environmental Monitoring and Assessment* 124(1-3): 167-194.

Johnson, O.W., M.H. Ruckelshaus, W.S. Grant, F.W. Waknitz, A.M. Garrett, G.J. Bryant, K. Neely, and J.J. Hard. 1999. Status Review of Coastal Cutthroat Trout from Washington, Oregon, and California. Seattle, Washington: Northwest Fisheries Science Center.

Johnson, O.W., W.S. Grant, R.G. Kope, K. Neely, F.W. Waknitz, and R.S. Waples. 1997. Status Review of Chum Salmon from Washington, Oregon, and California, NOAA Technical Memorandum Seattle, Washington: U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Johnson, S.L., and J.A. Jones. 2000. Stream Temperature Responses to Forest Harvest and Debris Flows in Western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences* 57(S2): 30-39.

Johnson, S.W., and J.F. Thedinga. 2005. Fish Use and Size of Eelgrass Meadows in Southeastern Alaska: A Baseline for Long-Term Assessment of Biotic Change. *Northwest Science* 79(2-3): 141-155.

Johnston, S.G., P.G. Slavich, and P. Hirst. 2005a. The Impact of Controlled Tidal Exchange on Drainage Water Quality in Acid Sulphate Soil Backswamps. *Agricultural Water Management* 73(2): 87-111.

Johnston, S.G., P.G. Slavich, and P. Hirst. 2005b. Opening Floodgates in Coastal Floodplain Drains: Effects on Tidal Forcing and Lateral Transport of Solutes in Adjacent Groundwater. *Agricultural Water Management* 74(1): 23-46.

Jones & Stokes. 2006. Overwater Structures and Non Structural Piling (White Paper). Prepared by Jones and Stokes Associates, in association with Anchor Environmental, L.L.C., and R2 Consultants for Washington Department of Fish and Wildlife, Olympia, Washington.

Jones, J.B., S.G. Fisher, and N.B. Grimm. 1995. Nitrification in the Hyporheic Zone of a Desert Stream Ecosystem. *Journal of the North American Benthological Society* 14: 249-258.

Joy, M.K., and R.G. Death. 2001. Control of Freshwater Fish and Crayfish Community Structure in Taranaki, New Zealand: Dams, Diadromy or Habitat Structure? *Freshwater Biology* 46(3): 417-429.

Kail, J. 2003. Influence of Large Woody Debris on the Morphology of Six Central European Streams. *Geomorphology* 51(1-3): 207-223.

Kaller, M.D., and W.E. Kelso. 2007. Association of Macroinvertebrate Assemblages with Dissolved Oxygen Concentration and Wood Surface Area in Selected Subtropical Streams of the Southeastern USA. *Aquatic Ecology* V41(1): 95-110.

Kalmijn, A.J. 1988. Hydrodynamic and Acoustic Field Detection. In *Sensory Biology of Aquatic Animals*, edited by J. Atema, R.R. Fay, A.N. Popper and W.N. Tavolga. New York: Springer-Verlag. pp. 83-130.

Karr, J.R. 1991. Biological Integrity - a Long-Neglected Aspect of Water-Resource Management. *Ecological Applications* 1(1): 66-84.

Katano, O., T. Nakamura, S. Abe, S. Yamamoto, and Y. Baba. 2006. Comparison of Fish Communities between Above- and Below-Dam Sections of Small Streams; Barrier Effect to Diadromous Fishes. *Journal of Fish Biology* 68(3): 767-782.

Kawamata, S. 2001. Adaptive Mechanical Tolerance and Dislodgement Velocity of the Kelp *Laminaria Japonica* in Wave-Induced Water Motion. *Marine Ecology-Progress Series* 211: 89-104.

Kelsey, K.A., and S.D. West. 1998. Riparian Wildlife. In *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*, edited by R.J. Naiman and R.E. Bilby. New York: Springer-Verlag.

Kendall, A.W., and W.H. Lenarz. 1986. Status of Early Life History Studies of Northeast Pacific Rockfishes. In *Proceedings International Rockfish Symposium*, Anchorage, Alaska: Alaska Sea Grant College Program. pp. 99-128.

Kingsford, R.T. 2000. Ecological Impacts of Dams, Water Diversions and River Management on Floodplain Wetlands in Australia. *Austral Ecology* 25(2): 109-127.

Kiorboe, T., F. Mohlenberg, and O. Nohr. 1981. Effect of Suspended Bottom Material on Growth and Energetics in *Mytilus-Edulis*. *Marine Biology* 61(4): 283-288.

Kishi, D., M. Murakami, S. Nakano, and Y. Taniguchi. 2004. Effects of Forestry on the Thermal Habitat of Dolly Varden (*Salvelinus Malma*). *Ecological Research* 19(3): 283-290.

Kitano, S., and K. Shimazaki. 1995. Spawning Habitat and Nest Depth of Female Dolly-Varden *Salvelinus-Malma* of Different Body-Size. *Fisheries Science* 61(5): 776-779.

Knudsen, E.E., and S.J. Dilley. 1987. Effects of Riprap Bank Reinforcement on Juvenile Salmonids in Four Western Washington Streams. *North American Journal of Fisheries Management* 7: 351-356.

Knudsen, F.R., P.S. Enger, and O. Sand. 1992. Awareness Reactions and Avoidance Responses to Sound in Juvenile Atlantic Salmon, *Salmo Salar*. *Journal of Fish Biology* 40: 523-534.

Knutson, K.L., and V.L. Naef. 1997. Management Recommendations for Washington's Priority Habitats. Riparian. Olympia, Washington: Washington State Department of Fish and Wildlife.

Koch, E.M. 2001. Beyond Light: Physical, Geological, and Geochemical Parameters as Possible Submersed Aquatic Vegetation Habitat Requirements. *Estuaries* 24(1): 1-17.

Kock, T.J., J.L. Congleton, and P.J. Anders. 2006. Effects of Sediment Cover on Survival and Development of White Sturgeon Embryos. *North American Journal of Fisheries Management* 26(1): 134-141.

Komar, P.D. 1998. *Beach Processes and Sedimentation*. Princeton, New Jersey: Prentice Hall.

Kondolf, G.M. 1997. Hungry Water: Effects of Dams and Gravel Mining on River Channels. *Environmental Management* 21(4): 533-551.

Kondolf, G.M., and M.G. Wolman. 1993. The Sizes of Salmonid Spawning Gravels. *Water Resources Research* 29(7): 2275-2285.

Kondolf, G.M., and R.R. Curry. 1986. Channel Erosion Along the Carmel River, Monterey County, California. *Earth Surface Processes and Landforms* 11: 307-319.

Kondolf, M.G., M.J. Sale, and M.G. Wolman. 1993. Modification of Fluvial Gravel Size by Spawning Salmonids. *Water Resources Research* 29(7): 2265-2274.

Konrad, C.P. 2000. The Frequency and Extent of Hydrologic Disturbances in Streams in the Puget Lowland, Washington. Ph.D. Dissertation Thesis, University of Washington, Seattle, Washington, 212 pp.

Kozlowski, T.T. 2002. Physiological-Ecological Impacts of Flooding on Riparian Forest Ecosystems. *Wetlands* 22(3): 550-561.

Kramer, D. E. and O’Connell, V.M., 1995. Guide to Northeast Pacific rockfishes: Genera *Sebastes* and *Sebastolobus*. *Alaska Sea Grant Marine Advisory Bulletin*, 25. 78pp.

Krueger, K., P. Chapman, M. Hallock, and T. Quinn. 2007. Some Effects of Suction Dredge Placer Mining on the Short-Term Survival of Freshwater Mussels in Washington Final Report (Draft) USFSW HPA/HCP Grant E-29-HP. Olympia, Washington: Washington Department of Fish and Wildlife Habitat and Fisheries Programs.

Kynard, B., E. Parker, and T. Parker. 2005. Behavior of Early Life Intervals of Klamath River Green Sturgeon, *Acipenser Medirostris* with a Note on Body Color. *Environmental Biology of Fishes* 72(1): 85-97.

Lagasse, P.F., L.W. Zevenbergen, J.D. Schall, and P.E. Clopper. 2001. Bridge Scour and Stream Instability Countermeasures. Washington D.C. Federal Highway Administration, U.S. Department of Transportation.

Lake, R.G., and S.G. Hinch. 1999. Acute Effects of Suspended Sediment Angularity on Juvenile Coho Salmon (*Oncorhynchus Kisutch*). *Canadian Journal of Fisheries and Aquatic Sciences* 56(5): 862-867.

Lamb, M.P., E. D'Asaro, and J.D. Parsons. 2004. Turbulent Structure of High-Density Suspensions Formed under Waves. *Journal of Geophysical Research-Oceans* 109(C12).

Lamberti, G.A., S.V. Gregory, L.R. Ashkenas, A.D. Steinman, and C.D. McIntire. 1989. Productive Capacity of Periphyton as a Determinant of Plant Herbivore Interactions in Streams. *Ecology* 70(6): 1840-1856.

Lane, E.W. 1955. The Importance of Fluvial Morphology in Hydraulic Engineering. *Proceedings of the American Society of Civil Engineers* 81: 745-761.

Langer, O.E., B.G. Shepherd, and P.R. Vroom. 1977. Biology of the Nass River Eulachon (*Thaleichthys Pacificus*). Department of Fisheries and Environment Canada, Fisheries and Marine Service.

Larinier, M. 1998. Small-Scale Hydropower Schemes and Migratory Fish Passage. *Houille Blanche-Revue Internationale De L Eau* 53(8): 46-51.

Laroche, W.A., and S.L. Richardson. 1981. Development of Larvae and Juveniles of the Rockfishes *Sebastes Entomelas* and *S. Zacentrus* (Family Scorpaenidae) and Occurrence Off Oregon, with Notes on Head Spines of *S. Mystinus*, *S. Flavidus*, and *S. Melanops*. *Fishery Bulletin* 79: 231-256.

Larsen, E.M., E. Rodrick, and R. Milner. 1995. Management Recommendations for Washington's Priority Species. Volume I: Invertebrates. Olympia, Washington: Washington Department of Fish and Wildlife.

Larson, K.W., and C.E. Moehl. 1990. Entrainment of Anadromous Fish by Hopper Dredge at the Mouth of the Columbia River. September 8-9, 1990. pp. 102-112.

Laughlin, J. 2004. Underwater Sound Levels Associated with the Construction of the SR 240 Bridge on the Yakima River at Richland. Seattle, Washington: Washington State Department of Transportation, Office of Air Quality and Noise.

Laughlin, J. 2005. Underwater Sound Levels Associated with the Restoration of the Friday Harbor Ferry Terminal. Seattle, Washington: Washington State Department of Transportation.

Laughlin, J. 2006. Underwater Sound Levels Associated with Pile Driving at the Cape Disappointment Boat Launch Facility, Wave Barrier Project. Olympia, Washington: Washington State Department of Transportation.

Lawler, D.M. 2005. Turbidity and Nephelometry. In *Encyclopedia of Analytical Science*, edited by P.J. Worsfold, A. Townshend, and C.F. Poole. Elsevier. pp. 343-351.

Leary, R.F., and F.W. Allendorf. 1997. Genetic Confirmation of Sympatric Bull Trout and Dolly Varden in Western Washington. *Transactions of the American Fisheries Society* 126: 715-20.

Lefebvre, S., P. Marmonier, G. Pinay, O. Bour, L. Aquilina, and J. Baudry. 2005. Nutrient Dynamics in Interstitial Habitats of Low-Order Rural Streams with Different Bedrock Geology. *Archiv Fur Hydrobiologie* 164(2): 169-191.

Lemieux, J.P. 2004. Proceedings of the DFO/PSAT Sponsored Marine Riparian Experts Workshop. Tsawwassen, British Columbia: Canadian Manuscript Report of Fisheries and Aquatic Sciences.

Leopold, L.B., M.G. Wolman, and J.P. Miller. 1964. *Fluvial Processes in Geomorphology*. New York: Freeman.

Levings, C.D., and G. Jamieson. 2001. Marine and Estuarine Riparian Habitats and their Role in Coastal Ecosystems, Pacific Region. Canadian Science Advisory Secretariat.

Lewis, A.F.J., M.D. McGurk, and M.G. Galesloot. 2002. Alcan's Kemano River Eulachon (*Thaleichthys Pacificus*) Monitoring Program 1988-1998. Kitimat, British Columbia: Consultant's report prepared by Ecofish Research Ltd. for Alcan Primary Metal Ltd.

Li, H.W., C.B. Schreck, and R.A. Tubb. 1984. Comparison of Habitat near Spur Dikes, Continuous Revetments, and Natural Banks for Larval, Juvenile, and Adult Fishers of the Willamette River. Corvallis, Oregon: U.S. Geological Survey, Water Resources Research Institute

Liang, H.Z., M.P. Lamb, and J.D. Parsons. 2007. Formation of a Sandy Near-Bed Transport Layer from a Fine-Grained Bed under Oscillatory Flow. *Journal of Geophysical Research-Oceans* 112(C2).

Liknes, G.A., and P.J. Graham. 1988. Westslope Cutthroat Trout in Montana: Life History, Status and Management. In *Status and Management of Cutthroat Trout*, edited by R.E. Gresswell. Bethesda, Maryland: American Fisheries Society. pp. 53-60.

Lisle, T.E., and J. Lewis. 1992. Effects of Sediment Transport on Survival of Salmonid Embryos in a Natural Stream - a Simulation Approach. *Canadian Journal of Fisheries and Aquatic Sciences* 49(11): 2337-2344.

Lister, D.B., R.J. Beniston, R. Kellerhals, and M. Miles. 1995. Rock Size Affects Juvenile Salmonid Use of Streambank Riprap. In *River, Coastal and Shoreline Protection: Erosion Control Using Riprap and Armourstone*, edited by C.R. Thorne, S.R. Abt, F.B.J. Barends, S.T. Maynord and K.W. Pilarczyk. John Wiley & Sons, Ltd. pp. 621-632.

Liu, H.Y., S.K. Zhang, Z.F. Li, X.G. Lu, and Q. Yang. 2004. Impacts on Wetlands of Large-Scale Land-Use Changes by Agricultural Development: The Small Sanjiang Plain, China. *Ambio* 33(6): 306-310.

Livingston, P.A. 1991. Food Habits and Population Level Consumption of Groundfish. In *Groundfish Food Habits and Predation on Commercially Important Prey Species in the Eastern Bering Sea from 1984 to 1986*, edited by P.A. Livingston. Seattle, Washington: U.S. Department of Commerce, NOAA Tech. Memo., NMFS F/NWC-207. pp. 9-88.

Lloyd, D.S. 1987. Turbidity as a Water Quality Standard for Salmonid Habitats in Alaska. *North American Journal of Fisheries Management* 7: 34-45.

Love, M.S., P. Morris, M. McCrae, and R. Collins. 1990. Life History Aspects of 19 Rockfish Species (Scorpaenidae: Sebastes) from the Southern California Bight. National Oceanic and Atmospheric Administration.

Lunz, J. 1985. An Analysis of Available Information Concerning the Entrainment of Oyster Larvae During Hydraulic Cutterhead Dredging Operations with Commentary on the Reasonableness of Seasonally Restricting Dredging Windows. Vicksburg, Mississippi: U.S. Army Corps of Engineers Waterways Experiment Station.

MacBroom, J.R. 1998. *The River Book - The Nature of Streams in Glaciated Terrains*. Hartford, Connecticut: Connecticut Department of Environmental Protection.

MacDonald, J.S., E.A. MacIsaac, and H.E. Herunter. 2003. The Effect of Variable-Retention Riparian Buffer Zones on Water Temperatures in Small Headwater Streams in Sub-Boreal Forest Ecosystems of British Columbia. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere* 33(8): 1371-1382.

Magilligan, F.J., and K.H. Nislow. 2005. Changes in Hydrologic Regime by Dams. *Geomorphology* 71(1-2): 61-78.

Mamelona, J., and T. Pelletier. 2003. Butyltins Biomagnification from Macroalgae to Green Sea Urchin: A Field Assessment. *Applied Organometallic Chemistry* 17(10): 759-766.

Marine, K.R., and J.J. Cech. 2004. Effects of High Water Temperature on Growth, Smoltification, and Predator Avoidance in Juvenile Sacramento River Chinook Salmon. *North American Journal of Fisheries Management* 24(1): 198-210.

Marsden, J.E., and M.A. Chotkowski. 2001. Lake Trout Spawning on Artificial Reefs and the Effect of Zebra Mussels: Fatal Attraction? *Journal of Great Lakes Research* 27(1): 33-43.

Marshall, D.W., M. Otto, J.C. Panuska, S.R. Jaeger, D. Sefton, and T.R. Baumberger. 2006. Effects of Hypolimnetic Releases on Two Impoundments and Their Receiving Streams in Southwest Wisconsin. *Lake and Reservoir Management* 22(3): 223-232.

Maser, C., and J. Sedell. 1994. *From the Forest to the Sea: The Ecology of Wood in Streams, Rivers, Estuaries, and Oceans*. Delray Beach, Florida: St. Lucie Press.

Matthews, K.R. 1990. An Experimental Study of the Habitat Preferences and Movement Patterns of Copper, Quillback, and Brown Rockfishes (*Sebastes* Spp.). *Environmental Biology of Fishes* 29: 161-178.

May, C.W. 2003. Stream-Riparian Ecosystems in the Puget Sound Lowland Eco-Region: A Review of Best Available Science. Watershed Ecology LLC.

Mayer, P., S. Reynolds, T.J. Canfield, and M.D. McCutchen. 2005. Riparian Buffer Width, Vegetative Cover, and Nitrogen Removal Effectiveness: A Review of Current Science and Regulations. EPA/600/R-05/118. Ada, Oklahoma: U.S. Environmental Protection Agency.

Mayfield, R.B., and J.J. Cech. 2004. Temperature Effects on Green Sturgeon Bioenergetics. *Transactions of the American Fisheries Society* 133(4): 961-970.

Mazur, M.M., and D.A. Beauchamp. 2003. A Comparison of Visual Prey Detection among Species of Piscivorous Salmonids: Effects of Light and Low Turbidities. *Environmental Biology of Fishes* 67(4): 397-405.

MBC Applied Environmental Sciences. 1987. Ecology of Important Fisheries Species Offshore California. MMS 86-0093. Washington, D.C. Pacific Outer Continental Shelf Region.

McCool, W.W., and J.D. Parsons. 2004. Sedimentation from Buoyant Fine-Grained Suspensions. *Continental Shelf Research* 24(10): 1129-1142.

McCormick, S.D., R.A. Cunjak, B. Dempson, M.F. O'Dea, and J.B. Carey. 1999. Temperature-Related Loss of Smolt Characteristics in Atlantic Salmon (*Salmo Salar*) in the Wild. *Canadian Journal of Fisheries and Aquatic Sciences* 56(9): 1649-1658.

McCullough, M.C., S. Spalding, and D. Sturdevant. 2001. Summary of Technical Literature Examining the Physiological Effects of Temperature on Salmonids. EPA-910-D-01-005. Washington, DC: U.S. Environmental Protection Agency.

McFarlane, G.A., and R.J. Beamish. 1986. Biology and Fishery of Pacific Hake *Merluccius Productus* in the Strait of Georgia. *International North Pacific Fisheries Commission Bulletin 50*: 365-392.

McGraw, K.A., and D.A. Armstrong. 1990. Fish Entrainment by Dredges in Grays Harbor, Washington. *Effects of Dredging on Anadromous Pacific Coast Fishes*, pp. 113-131.

McIntosh, M.D., M.E. Benbow, and A.J. Burky. 2002. Effects of Stream Diversion on Riffle Macroinvertebrate Communities in a Maui, Hawaii, Stream. *River Research and Applications* 18(6): 569-581.

McKinnell, S., J.J. Pella, and M.L. Dahlberg. 1997. Populations-Specific Aggregations of Steelhead Trout (*Oncorhynchus Mykiss*) in the North Pacific Ocean. *Canadian Journal of Fisheries and Aquatic Sciences* 54: 2368-2376.

McLeay, D.L., I. Birtwell, G. Hartman, and G. Ennis. 1987. Response of Arctic Grayling (*Thymallus Arcticus*) to Acute and Prolonged Exposure to Yukon Placer Mining Sediment. *Canadian Journal of Fisheries and Aquatic Sciences* 44: 658-673.

McMichael, G.A., L. Fritts, and T.N. Pearsons. 1998. Electrofishing Injury to Stream Salmonids; Injury Assessment at the Sample, Reach, and Stream Scales. *North American Journal of Fisheries Management* 18: 894-904.

Meadows, G.A., S.D. Mackey, R.R. Goforth, D.M. Mickelson, T.B. Edil, J. Fuller, D.E. Guy, L.A. Meadows, and E. Brown. 2005. Cumulative Habitat Impacts of Nearshore Engineering. *Journal of Great Lakes Research* 31: 90-112.

Meeuwig, M.H., J.M. Bayer, and J.G. Seelye. 2005. Effects of Temperature on Survival and Development of Early Life Stage Pacific and Western Brook Lampreys. *Transactions of the American Fisheries Society* 134(1): 19-27.

Meier, A.H., and N.D. Horseman. 1977. Stimulation and Depression of Growth, Fat Storage, and Gonad Weight by Daily Stimulus in the Teolost Fish, *Tilapia Aurea*. World Mariculture Society, *Eighth Annual Meeting.*

Meldgaard, T., E.E. Nielsen, and V. Loeschcke. 2003. Fragmentation by Weirs in a Riverine System: A Study of Genetic Variation in Time and Space among Populations of European Grayling (*Thymallus Thymallus*) in a Danish River System. *Conservation Genetics* 4(6): 735-747.

Michny, F., and R. Deibel. 1986. Sacramento River Chico Landing to Red Bluff Project 1985 Juvenile Salmon Study. Sacramento, California: U.S. Army Corps of Engineers.

Mickett, J.B., M.C. Gregg, and H.E. Seim. 2004. Direct Measurements of Diapycnal Mixing in a Fjord Reach - Puget Sound's Main Basin. *Estuarine Coastal and Shelf Science* 59(4): 539-558.

Millar, R.G., and M.C. Quick. 1998. Stable Width and Depth of Gravel-Bed Rivers with Cohesive Banks. *Journal of Hydraulic Engineering-ASCE* 124(10): 1005-1013.

Miller, B.S., C.A. Simenstad, and L.R. Moulton. 1976. Puget Sound Baseline Program: Nearshore Fish Survey, Annual Report July 1974-September 1975. Report No. 76-04. Seattle, Washington: Fisheries Research Institute, University of Washington.

Miller, D.R., R.K.L. Emmett, and S.A. Hinton. 1990. A Preliminary Survey of Benthic Invertebrates in the Vicinity of the Coos Bay Oregon, Navigation Channel: Coastal Zone and Estuarine Studies. Prepared for U.S. Army Corps of Engineers by Northwest Fisheries Science Center, National Marine Fisheries Center, Seattle, Washington.

Miller, M.C., I.N. McCave, and P.D. Komar. 1977. Threshold of Sediment Motion under Unidirectional Currents. *Sedimentology* 24(4): 507-527.

Miller, M.C., R.M. Thom, G.D. Williams, J.A. Southard, S.L. Blanton, and L.K. O'Rourke. 2001. Effects of Shoreline Hardening and Shoreline Protection Features on Fish Utilization and Behavior, Washaway Beach, Washington. Richland, Washington: Pacific Northwest National Laboratory.

Miller, S.W., D. Wooster, and J. Li. 2007. Resistance and Resilience of Macroinvertebrates to Irrigation Water Withdrawals. *Freshwater Biology* 52: 2494-2510.

Mills, K.E., and M.S. Fonseca. 2003. Mortality and Productivity of Eelgrass *Zostera Marina* under Conditions of Experimental Burial with Two Sediment Types. *Marine Ecology-Progress Series* 255: 127-134.

Minakawa, N., and G.F. Kraft. 2005. Homing Behavior of Juvenile Coho Salmon (*Oncorhynchus Kisutch*) within an Off-Channel Habitat. *Ecology of Freshwater Fish* 14(2): 197-201.

Misitano, D.A., E. Casillas, and C.R. Haley. 1994. Effects of Contaminated Sediments on Viability, Length, DNA and Protein-Content of Larval Surf Smelt, *Hypomesus-Pretiosus*. *Marine Environmental Research* 37(1): 1-21.

Mohlenberg, F., and T. Kiorboe. 1981. Growth and Energetics in *Spisula-Subtruncata* (Da Costa) and the Effect of Suspended Bottom Material. *Ophelia* 20(1): 79-90.

Mongillo, P.E., and M. Hallock. 1998. Washington State Status Report for the Margined Sculpin. Washington Department of Fish and Wildlife.

Mongillo, P.E., and M. Hallock. 1999. Washington State Status Report for the Olympic Mudminnow. Olympia, Washington: Washington Department of Fish and Wildlife.

Montgomery, D.R., E.M. Beamer, G.R. Pess, and T.P. Quinn. 1999. Channel Type and Salmonid Spawning Distribution and Abundance. *Canadian Journal of Fisheries and Aquatic Science* 56: 377-387.

Montgomery, D.R., J.M. Buffington, N.P. Peterson, D. Schuett-Hames, and T.P. Quinn. 1996. Stream-Bed Scour, Egg Burial Depths, and the Influence of Salmonid Spawning on Bed Surface Mobility and Embryo Survival. *Canadian Journal of Fisheries and Aquatic Sciences* 53: 1061-1070.

Moore, H.L., and H.W. Newman. 1956. Effects of Sound Waves on Young Salmon. Special Science Report-Fisheries 172. U.S. Fish and Wildlife Service, 19p.

Moore, K.A. 2004. Influence of Seagrasses on Water Quality in Shallow Regions of the Lower Chesapeake Bay. *Journal of Coastal Research* 20: 162-178.

Morioka, T., and H. Kuwada. 2002. The Upper Limit of Inhabiting Temperature and the Diet of Juvenile Pacific Cod *Gadus Macrocephalus* in the Northern Part of Nanao Bay and Its Vicinity, Japan. *Nippon Suisan Gakkaishi* 68(3): 345-350.

Morton, J.W. 1977. Ecological Effects of Dredging and Dredge Spoil Disposal: A Literature Review. Technical Paper 94. U.S. Fish and Wildlife Service, 33 p.

Moser, H.G., N.C.H. Lo, and P.E. Smith. 1997. Vertical Distribution of Pacific Hake Eggs in Relation to Stage of Development and Temperature. *California Cooperative Oceanic Fisheries Investigations Reports* 38: 120-126.

Moser, M.L., P.A. Ocker, L.C. Stuehrenberg, and T.C. Bjornn. 2002. Passage Efficiency of Adult Pacific Lampreys at Hydropower Dams on the Lower Columbia River, USA. *Transactions of the American Fisheries Society* 131(5): 956-965.

Moulton, L.L. 1977. Ecological Analysis of Fishes Inhabiting the Rocky Nearshore Regions of Northern Puget Sound. Ph.D. Dissertation Thesis, University of Washington.

Moyle, P.B., and J.J. Cech. 1988. *Fishes: An Introduction to Ichthyology*. Second Edition. New Jersey: Prentice Hall Publishing.

Moyle, P.G. 1976. Fish Introduction in California: History and Impact on Native Fishes. *Biological Conservation* 9: 101-118.

Mulholland, P.J., E.R. Marzolf, J.R. Webster, D.R. Hart, and S.P. Hendricks. 1997. Evidence That Hyporheic Zones Increase Heterotrophic Metabolism and Phosphorus Uptake in Forest Streams. *Limnology and Oceanography* 42: 443-451.

Mulholland, P.J., J.D. Newbold, J.W. Elwood, L.A. Ferren, and J.R. Webster. 1985. Phosphorus Spiraling in a Woodland Stream: Seasonal Variations. *Ecology* 66: 1012-1023.

Mulholland, R. 1984. Habitat Suitability Index Models: Hard Clam. National Coastal Ecosystems Team. Division of Biological Services Research and Development, Fish and Wildlife Service, U.S. Department of the Interior.

Mulliss, R., D.M. Revitt, and R.B.E. Shutes. 1997. The Impacts of Discharges from Two Combined Sewer Overflows on the Water Quality of an Urban Watercourse. *Water Science and Technology* 36(8-9): 195-199.

Murakami, K., and K. Takeishi. 1977. Behavior of Heavy Metals and PCBs in Dredging and Treating of Bottom Deposits. In *Management of Bottom Sediments Containing Toxic Substances*, edited by S.A. Peterson and K.K. Randolph. Washington, DC: Proceedings of the 2nd US/Japan Experts Meeting USEPA-600/3-77-083. pp. 26-42.

Murphy, M. 1998. Primary Productivity. In *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*, edited by R.J. Naiman and R.E. Bilby. New York: Springer-Verlag. pp. 144-168.

Murphy, M.L., and W.R. Meehan. 1991. Stream Ecosystems. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*, edited by W.R. Meehan. Bethesda, Maryland: American Fisheries Society Special Publication 19. pp. 17-46

Murphy, M.L., C.P. Hawkins, and N.H. Anderson. 1981. Effects of Canopy Modification and Accumulated Sediment on Stream Communities. *Transactions of the American Fisheries Society* 110(4): 469-478.

Myers, J.M., R.G. Kope, G.J. Bryant, D. Teel, L.J. Lierheimer, T.C. Wainwright, W.S. Grand, F.W. Waknitz, K. Neely, S.T. Lindley, and R.S. Waples. 1998. Status Review of Chinook Salmon from Washington, Idaho, Oregon, and California. Seattle, Washington: Northwest Fisheries Science Center.

Myers, R.D. 1993. Slope Stabilization and Erosion Control Using Vegetation: A Manual of Practice for Coastal Property Owners. Olympia, Washington: Shorelands and Coastal Zone Management Program, Washington Department of Ecology.

Myrberg, A.A. 1972. Using Sound to Influence the Behavior of Free-Ranging Marine Animals. *Plenum* 2: 435-368.

Myrberg, A.A., and R.J. Riggio. 1985. Acoustically Mediated Individual Recognition by a Coral Reef Fish (*Pomacentrus Partitus*). *Animal Behavior* 33: 411-416.

Naiman, R.J., E.V. Balian, K.K. Bartz, R.E. Bilby, and J.J. Latterell. 2002. Dead Wood Dynamics in Stream Ecosystems. USDA Forest Service.

Naiman, R.J., T.J. Beechie, L.E. Benda, D.R. Berg, P.A. Bisson, L.H. MacDonald, M.D. O'Connor, P.L. Olson, and E.A. Steel. 1992. Fundamental Elements of Ecologically Healthy Watersheds in the Pacific Northwest Coastal Ecoregion. In *Watershed Management – Balancing Sustainability and Environmental Change*, edited by R.J. Naiman. New York: Springer Verlag.

Nakamoto, R.J., and T.T. Kisanuki. 1995. Age and Growth of Klamath River Green Sturgeon (*Acipenser Medirostris*). Project #93-FP-13. Arcata, California: U.S. Forest Service.

Nakayama, T., M. Watanabe, K. Tanji, and T. Morioka. 2007. Effect of Underground Urban Structures on Eutrophic Coastal Environment. *Science of the Total Environment* 373(1): 270-288.

National Conservation Training Center. 2004. The Analytical Approach to Consultation. Lacey, Washington: Advanced Interagency Consultation - Regional training curriculum.

Nedeau, E., A.K. Smith, and J. Stone. 2005. Freshwater Mussels of the Pacific Northwest. Vancouver, Washington: U.S. Fish and Wildlife Service.

Nedwell, J., A. Martin, and N. Mansfield. 1993. Underwater Tool Noise: Implications for Hearing Loss. In *Advances in Underwater Technology*, *Ocean Science and Offshore Engineering*, edited by Subtech '93. Dordrecht, The Netherlands: Kluwer Academic Publishers. pp. 267-275.

Nedwell, J., A. Turnpenny, J. Langworthy, and B. Edwards. 2003. Measurements of Underwater Noise During Piling at the Red Funnel Terminal, Southampton, and Observations of Its Effect on Caged Fish. Subacoustics LTD.

Nedwell, J., and B. Edwards. 2002. Measurements of Underwater Noise in the Arun River During Piling at County Wharf, Littlehampton. *Subacoustech*.

Neitzel, D.A. and T.J. Frest. 1990. Survey of Columbia River Basin Streams for Columbia Pebblesnail and Shortface Lanx. *Fisheries* 15(2):2-3.

Neitzel, D.A., and T.J. Frest. 1989. Survey of Columbia River Basin Streams for Giant Columbia River Spire Snail, *Fluminicola Columbiana* and Great Columbia River Limpet, *Fisherola Nuttalli*. Richland, Washington: Pacific Northwest Laboratory.

Nelson, D.R. 1965. Hearing and Acoustic Orientation in the Lemon Shark *Negaprion Brevirostris* (Poey), and Other Large Sharks. *Bulletin of Southern Californian Academic Sciences* 68(3): 131-137.

Nelson, D.R., R.H. Johnson, and L.G. Waldrop. 1969. Responses in Bahamian Sharks and Groupers to Low-Frequency, Pulsed Sounds. *Bulletin of Southern Californian Academic Sciences* 68:131-137.

Nelson, T.A., A.V. Nelson, and M. Tjoelker. 2003. Seasonal and Spatial Patterns Of "Green Tides" (Ulvoid Algal Blooms) and Related Water Quality Parameters in the Coastal Waters of Washington State, USA. *Botanica Marina* 46(3): 263-275.

Neraas, L.P., and P. Spruell. 2001. Fragmentation of Riverine Systems: The Genetic Effects of Dams on Bull Trout (*Salvelinus Confluentus)* in the Clark Fork River System. *Molecular Ecology* 10(5): 1153-64.

Newbold, S.C., and R. Iovanna. 2007. Population level impacts of cooling water withdrawals on harvested fish stocks. *Environmental Science and Technology* 41 (7): 2108-2114.

Newcomb, T.W., and T.A. Flagg. 1983. Some Effects of Mt. St.-Helens Volcanic Ash on Juvenile Salmon Smolts. *Marine Fisheries Review* 45(2): 8-12.

Newcombe, C.P., and D.D. MacDonald. 1991. Effects of Suspended Sediments on Aquatic Ecosystems. *North American Journal of Fisheries Management* 11: 72-82.

Newcombe, C.P., and J.O.T. Jensen. 1996. Channel Suspended Sediment and Fisheries: A Synthesis for Quantitative Assessment of Risk and Impact. *North American Journal of Fisheries Management* 16(693-727).

Newell, R.C., L.J. Seiderer, N.M. Simpson, and J.E. Robinson. 2004. Impacts of Marine Aggregate Dredging on Benthic Macrofauna Off the South Coast of the United Kingdom. *Journal of Coastal Research* 20(1): 115-125.

Nightingale, B., and C. Simenstad. 2001a. Dredging Activities: Marine Issues. Prepared for the Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation by Wetland Ecosystem Team, University of Washington, Seattle, Washington.

Nightingale, B., and C. Simenstad. 2001b. Marine Overwater Structures: Marine Issues. Prepared for Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation by Wetland Ecosystem Team, University of Washington, Seattle, Washington.

Nittrouer, C.A., and R.W. Sternberg. 1975. Fate of a Fine-Grained Dredge Spoils Deposit in a Tidal Channel of Puget-Sound, Washington. *Journal of Sedimentary Petrology* 45(1): 160-170.

NMFS. 1990. West Coast of North America Coastal and Ocean Zones Strategic Assessment: Data Atlas. Washington, District of Columbia: NOAA's National Marine Fisheries Service.

NMFS. 2006. Endangered Species Act - Section 7 Formal Consultation Biological and Conference Opinion and Magnuson Stevens Fishery Management Conservation and Management Act Essential Fish Habitat Consultation Stream Crossing Structure Replacement and Removal Activities, Snake and Clearwater River Basins, 170601 & 170603, Idaho. National Marine Fisheries Service.

NMFS. 2007a. Species of Concern and Candidate Species: Pinto Abalone. Office of Protected Covered Species Paper. Seattle, Washington: National Marine Fisheries Service.

NMFS. 2007b. Rationale for the Use of 187 dB Sound Exposure Level for Pile Driving Impacts Threshold. Unpublished memorandum. Seattle, Washington: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Noggle, C.C. 1978. Behavioral, Physiological and Lethal Effects of Suspended Sediment on Juvenile Salmonids. Master’s Thesis, University of Washington, Seattle, Washington.

Nordstrom, K.F. 1992. *Estuarine Beaches: An Introduction of the Physical and Human Factors Affecting Use and Management of Beaches in Estuaries, Lagoons, Bays and Fjords*. New York: Elsevier Applied Science

NRC. 2001. Final Species Memorandum and Habitat Assessment in the King County HCP Planning Area. Volume 2: Marine Fish. Prepared for King County Wastewater Treatment Division by Natural Resources Consultants, Inc., Seattle, Washington. May 2001.

Nybakken, J.W., and M.D. Bertness. 2005. *Marine Biology: An Ecological Approach*. San Francisco: Pearson Benjamin Cummings.

O’Connell, V.M., and D.W. Carlile. 1993. Habitat-Specific Density of Adult Yelloweye Rockfish *Sebastes Ruberrimus* in the Eastern Gulf of Alaska. *Fishery Bulletin* 91: 304-309.

O'Connor, J.P., D.J. O'Mahony, J.M. O'Mahony, and T.J. Gienane. 2006. Some Impacts of Low and Medium Head Weirs on Downstream Fish Movement in the Murray-Darling Basin in Southeastern Australia. *Ecology of Freshwater Fish* 15(4): 419-427.

Olla, B.L., M.W. Davis, and C.B. Schreck. 1995. Stress-Induced Impairment of Predator Evasion and Non-Predator Mortality in Pacific Salmon. *Aquaculture Research* 26(6): 393-398.

Oullet, P., and J.J. Dodson. 1985. Tidal Exchanges of Anadromous Rainbow Smelt (*Osmerus Mordax*) Larvae between a Shallow Spawning Tributary and the St. Lawrence Estuary. *Canadian Journal of Fisheries and Aquatic Sciences* 42: 1352-1358.

Pacific Biodiversity Institute. 2007. Homepage. Available at: http://www.pacificbio.org/ (accessed July 2007).

Palmer, R.W., and J.H. Okeeffe. 1990. Downstream Effects of Impoundments on the Water Chemistry of the Buffalo River (Eastern Cape), South-Africa. *Hydrobiologia* 202(1-2): 71-83.

Paragamian, V.L., G. Kruse, and V. Wakkinen. 2001. Spawning Habitat of Kootenai River White Sturgeon, Post-Libby Dam. *North American Journal of Fisheries Management* 21(1): 22-33.

Parkhill, K.L., and J.S. Gulliver. 2002. Effect of Inorganic Sediment on Whole-Stream Productivity. *Hydrobiologia* 472(1-3): 5-17.

Paul, A.J., and J.M. Paul. 1998. Respiration Rate and Thermal Tolerances of Pinto Abalone *Haliotis Kamtschatkana*. *Journal of Shellfish Research* 17(3): 743-745.

Pauley, G.B., K.L. Oshima, and G.L. Thomas. 1988. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest)--Sea-Run Cutthroat Trout. U.S. Fish and Wildlife Service Biological Report.

Pearlstine, L.G., W.M. Kitchens, P.J. Latham, and R.D. Bartleson. 1993. Tide Gate Influences on a Tidal Marsh. *Water Resources Bulletin* 29(6): 1009-1019.

Penczak, T., G. Zieba, H. Koszalinski, and A. Kruk. 2003. The Importance of Oxbow Lakes for Fish Recruitment in a River System. *Archiv Fur Hydrobiologie* 158(2): 267-281.

Pentec Environmental, Inc. 1991. Port of Everett, Dredging Monitoring Surveys, 1990. Prepared for Port of Everett, Edmonds, Washington.

Penttila, D.E. 1978. Studies of the Surf Smelt (*Hypomesus Pretiosus*) in Puget Sound. Technical Report 42. Olympia, Washington: Washington Department of Fisheries.

Penttila, D.E. 1995. Investigations of the Spawning Habitat of the Pacific Sand Lance (*Ammodytes Hexapterus*) in Puget Sound. *Proceedings of Puget Sound Research 1995,* Seattle, Washington, pp. 855-859.

Penttila, D.E. 2000. Forage Fishes of the Puget Sound Region. *NWSC/PSAMP Data Conference*, LaConner, Washington.

Penttila, D.E. 2001. Effects of Overhanging Shading Vegetation on Egg Survival for Summer-Spawning Surf Smelt on Upper Intertidal Beaches in Northern Puget Sound, Washington, Draft. Washington Department of Fish and Wildlife, Marine Resources Division.

Peters, R.J., B.R. Missildine, and D.L. Low. 1998. Seasonal Fish Densities near River Banks Stabilized with Various Stabilization Methods; First Year Report of the Flood Technical Assistance Project. Lacey, Washington: U.S. Fish and Wildlife Service, North Pacific Coast Ecoregion.

Peterson, H.W.U., and L. Amiotte. 2006. Decline of Skokomish Nation Spot Shrimp Catch in Low Dissolved Oxygen Waters of the Hood Canal, Puget Sound, State of Washington. *Ethnicity & Disease* 16(4): 17-17.

Peterson, J.T., N.P. Banish, and R.F. Thurow. 2005. Are Block Nets Necessary? Movement of Stream-Dwelling Salmonids in Response to Three Common Survey Methods. *North American Journal of Fish Management* 25: 732-743.

Peterson, J.T., R.F. Thurow, and J.W. Guzevich. 2004. An Evaluation of Multipass Electrofishing for Estimating the Abundance of Stream-Dwelling Salmonids. *Transactions of the American Fisheries Society* 133(2): 462-475.

Peven, C.M. 1987. Downstream Migration Timing of Two Stocks of Sockeye Salmon on the Mid-Columbia River. *Northwest Science* 61(3): 186-190.

Phillips, R.C. 1984. Ecology of Eelgrass Meadows in the Pacific Northwest: A Community Profile. FWS/OBS-84/24. Portland, Oregon: U.S. Fish and Wildlife Service.

Pickett, P.J. 1997. Pollutant Loading Capacity for the Black River, Chehalis River System, Washington. *Journal of the American Water Resources Association* 33(2): 465-480.

Pihl, L., S. Baden, N. Kautsky, P. Ronnback, T. Soderqvist, M. Troell, and H. Wennhage. 2006. Shift in Fish Assemblage Structure Due to Loss of Seagrass *Zostera Marina* Habitats in Sweden. *Estuarine Coastal and Shelf Science* 67(1-2): 123-132.

Pillard, D.A. 1996. Assessment of Benthic Macroinvertebrate and Fish Communities in a Stream Receiving Storm Water Runoff from a Large Airport. *Journal of Freshwater Ecology* 11(1): 51-59.

Poole, G., J. Dunham, M. Hicks, D. Keenan, J. Lockwood, E. Materna, D. McCullough, C. Mebane, J. Risley, S. Sauter, S. Spaulding, and D. Sturdevant. 2001. Technical Synthesis Scientific Issues Relating to Temperature Criteria for Salmon, Trout, and Char Native to the Pacific. EPA 910-R-01-007. Environmental Protection Agency, Region 10.

Poole, G.C., and C.H. Berman. 2001a. An Ecological Perspective on in-Stream Temperature: Natural Heat Dynamics and Mechanisms of Human-Caused Thermal Degradation. *Environmental Management* 27(6): 787-802.

Poole, G.C., and C.H. Berman. 2001b. Pathways of Human Influence on Water Temperature Dynamics in Stream Channels. *Environmental Management* 27: 787-802.

Popper, A.N., and N.L. Clarke. 1976. The Auditory System of the Goldfish (*Carassius Auratus*): Effects of Intense Acoustic Stimulation. *Comparative Biochemistry and Physiology* 53: 11-18.

Popper, A.N., and R.R. Fay. 1973. Sound Detection and Processing by Teleost Fishes - Critical Review. *Journal of the Acoustical Society of America* 53(6): 1515-1529.

Popper, A.N., and R.R. Fay. 1993. Sound Detection and Processing by Fish - Critical-Review and Major Research Questions. *Brain Behavior and Evolution* 41(1): 14-38.

Popper, A.N., M.E. Smith, P.A. Cott, B.W. Hanna, A.O. MacGillivray, M.E. Austin, and D.A. Mann. 2005. Effects of Exposure to Seismic Airgun Use on Hearing of Three Fish Species. *Journal of the Acoustical Society of America* 117(6): 3958-3971.

Portnoy, J.W. 1991. Summer Oxygen Depletion in a Diked New-England Estuary. *Estuaries* 14(2): 122-129.

Power, M.E., W.E. Dietrich, and J.C. Finlay. 1996. Dams and Downstream Aquatic Biodiversity: Potential Food Web Consequences of Hydrologic and Geomorphic Change. *Environmental Management* 20(6): 887-895.

Pusch, M., D. Fiebig, I. Brettar, H. Eisenmann, B.K. Ellis, L.A. Kaplan, M.A. Lock, M.W. Naegeli, and W. Traunspurger. 1998. The Role of Micro-Organisms in the Ecological Connectivity of Running Waters. *Freshwater Biology* 40(3): 453-495.

Qiao, F.L., J. Ma, C.S. Xia, Y.Z. Yang, and Y.L. Yuan. 2006. Influences of the Surface Wave-Induced Mixing and Tidal Mixing on the Vertical Temperature Structure of the Yellow and East China Seas in Summer. *Progress in Natural Science* 16(7): 739-746.

Quinn, T.P. 2005. *The Behavior and Ecology of Pacific Salmon and Trout*. Seattle, Washington: University of Washington Press.

Quintella, B.R., N.O. Andrade, A. Koed, and P.R. Almeida. 2004. Behavioral Patterns of Sea Lampreys' Spawning Migration through Difficult Passage Areas, Studied by Electromyogram Telemetry. *Journal of Fish Biology* 65(4): 961-972.

Quirollo, L.F. 1992. Pacific Hake. In *California's Living Marine Resources and Their Utilization. California Sea Grant College Program*, edited by W.S. Leet, C.M. Dewees and C.W. Haugen. Davis, California. pp. 129.

Rabalais, N.N., R.E. Turner, and W.J. Wiseman. 2001. Hypoxia in the Gulf of Mexico. *Journal of Environmental Quality* 30: 320-329.

Rabalais, N.N., W.J. Wiseman, R.E. Turner, B.K. SenGupta, and Q. Dortch. 1996. Nutrient Changes in the Mississippi River and System Responses on the Adjacent Continental Shelf. *Estuaries* 19: 386-47.

Ram, J.L., P. Fong, R.P. Croll, S.J. Nichols, and D. Wall. 1992. The Zebra Mussel (*Dreissena-Polymorpha*), a New Pest in North-America - Reproductive Mechanisms as Possible Targets of Control Strategies. *Invertebrate Reproduction & Development* 22(1-3): 77-86.

Reilly, C.A., T.W. Wyllie-Echeverria, and S. Ralston. 1992. Interannual Variation and Overlap in the Diets of Pelagic Juvenile Rockfish (Genus: *Sebastes*) Off Central California. *Fishery Bulletin* 90: 505-515.

Reiman, B.E., and J.D. McIntyre. 1993. Demographic and Habitat Requirements for the Conservation of Bull Trout. Ogden, Utah: USDA Forest Service Intermountain Research Station.

Reine, K., and D. Clarke. 1998. Entrainment by Hydraulic Dredges - a Review of Potential Impacts. Technical Note. DOER-EI. U.S. Army Corps of Engineers, Engineer Research and Development Center.

Reyff, J., P. Donavan, and C.R. Greene Jr. 2003. Underwater Sound Levels Associated with Seismic Retrofit Construction of the Richmond-San Rafael Bridge. Prepared for California Department of Transportation by Illingworth & Rodkin, Inc. and Greeneridge Sciences Sacramento, California.

Rice, C.A. 2006. Effects of Shoreline Modification on a Northern Puget Sound Beach: Microclimate and Embryo Mortality in Surf Smelt. *Estuaries and Coasts* 29(1): 63-71.

Rice, C.E., and K.C. Kadavy. 1994. Riprap Design Downstream of Submerged Pipe Outlets. *Transactions of the ASAE* 37(1): 85-94.

Richard, J.D. 1968. Fish Attraction with Pulsed Low Frequency Sound. *Journal of the Fisheries Research Board of Canada* 25(7): 1441-1452.

Richardson, E.V., and S.R. Davis. 2001. Evaluating Scour at Bridges. Fourth Edition. FHWA NHI 01-001 HEC-18, Hydraulic Engineering Circular No. 19. Washington D.C: Federal Highway Administration, U.S. Department of Transportation.

Richardson, M.D., A.G. Carey, and W.A. Colgate. 1977. Aquatic Disposal Field Investigations - Columbia River Disposal Site, Oregon. Appendix C: The Effects of Dredged Material Disposal on Benthic Assemblages. Vicksburg, Mississippi: Report to U.S. Army Corps of Engineers Waterways Expt. Station.

Richter, A., and S.A. Kolmes. 2005. Maximum Temperature Limits for Chinook, Coho, and Chum Salmon, and Steelhead Trout in the Pacific Northwest. *Reviews in Fisheries Science* 13(1): 23-49.

Robinson, T.C., and J.M. Bayer. 2005. Upstream Migration of Pacific Lampreys in the John Day River, Oregon: Behavior, Timing, and Habitat Use. *Northwest Science* 79(2-3): 106-119.

Roegner, G.C., B.M. Hickey, J.A. Newton, A.L. Shanks, and D.A. Armstrong. 2002. Wind-Induced Plume and Bloom Intrusions into Willapa Bay, Washington. *Limnology and Oceanography* 47(4): 1033-1042.

Rolauffs, P., D. Hering, and S. Lohse. 2001. Composition, Invertebrate Community and Productivity of a Beaver Dam in Comparison to Other Stream Habitat Types. *Hydrobiologia* 459: 201-212.

Roman, C.T., W.A. Niering, and R.S. Warren. 1984. Salt Marsh Vegetation Change in Response to Tidal Restriction. *Environmental Management* 8: 141-150.

Rooper, C.N., D.R. Gunderson, and B.M. Hickey. 2006. An Examination of the Feasibility of Passive Transport from Coastal Spawning Grounds to Estuarine Nursery Areas for English Sole. *Estuarine Coastal and Shelf Science* 68(3-4): 609-618.

Rosenthal, R.J., V. Moran-O'Connell, and M.C. Murphy. 1988. Feeding Ecology of Ten Species of Rockfishes (Scorpaenidae) from the Gulf of Alaska. *California Fish and Game* 74: 16-36.

Rowe, D., J. Smith, B. Baillie, and M. Meleason. 2004. Wood in Streams: How Much Is Good for Fish? *Water and Atmosphere* 12(1): 16-17.

Rulifson, R.A., and B.L. Wall. 2006. Fish and Blue Crab Passage through Water Control Structures of a Coastal Bay Lake. *North American Journal of Fisheries Management* 26(2): 317-326.

Russel, I.C., A. Moore, S. Ives, L.T. Kell, M.J. Ives, and R.O. Stonehewer. 1998. The Migratory Behavior of Juvenile and Adult Salmonids in Relation to an Estuarine Barrage. *Hydrobiologia* 371/372: 321-333.

Saiki, M.K., and B.A. Martin. 2001. Survey of Fishes and Environmental Conditions in Abbotts Lagoon, Point Reyes National Seashore, California. *California Fish and Game* 87(4): 123-138.

Salo, E., N. Bax, T. Prinslow, C. Whitmus, B. Snyder, and C.A. Simenstad. 1980. The Effects of Construction of Naval Facilities on the Outmigration of Juvenile Salmonids from Hood Canal, Washington. Final Report FRI-UW-8006. Seattle, Washington: University of Washington, Fisheries Research Institute.

Salo, E.O. 1991. Life History of Chum Salmon (*Oncorhynchus Keta*). In *Pacific Salmon Life Histories*, edited by C. Groot and L. Margolis. Vancouver, British Columbia: University of British Columbia Press. pp. 231-310.

Sammut, J., I. White, and M.D. Melville. 1996. Acidification of an Estuarine Tributary in Eastern Australia Due to Drainage of Acid Sulfate Soils. *Marine and Freshwater Research* 47(5): 669-684.

Sampson, D.B. 1996. Stock Status of Canary Rockfish Off Oregon and Washington in 1996: Appendix C in Pacific Fishery Management Council. Status of the Pacific Coast Groundfish Fishery through 1996 and Recommended Acceptable Biological Catches for 1997: Stock Assessment and Fishery Evaluation. Portland, Oregon: Pacific Fishery Management Council.

Sand-Jensen, K. 1998. Influence of Submerged Macrophytes on Sediment Composition and Near-Bed Flow in Lowland Streams. *Freshwater Biology* 39(4): 663-679.

Scavia, D., and S.B. Bricker. 2006. Coastal Eutrophication Assessment in the United States. *Biogeochemistry* 79(1-2): 187-208.

Schaffter, R.G., P.A. Jones, and J.G. Karlton. 1983. Sacramento River and Tributaries Bank Protection and Erosion Control Investigation–Evaluation of Impacts on Fisheries. Sacramento, California: U.S. Army Corps of Engineers.

Schlesinger, W.H. 1997. *Biogeochemistry: An Analysis of Global Change*. San Diego, California: Academic Press.

Schmetterling, D.A., C.G. Clancy, and T.M. Brandt. 2001. Effects of Riprap Bank Reinforcement on Stream Salmonids in the Western United States. *Fisheries* 26(7): 6-13.

Scholik, A.R., and H.Y. Yan. 2001. Effects of Underwater Noise on Auditory Sensitivity of a Cyprinid Fish. *Hearing Research* 152: 17-24.

Scholik, A.R., and H.Y. Yan. 2002. The Effects of Noise on the Auditory Sensitivity of the Bluegill Sunfish, *Lepomis Macrochirus*. *Comparative Biochemistry and Physiology* A133: 43-52.

Schoning, R.W., and D.R. Johnson. 1956. A Measured Delay in the Migration of Adult Chinook Salmon at Bonneville Dam on the Columbia River. Contribution No. 23. Portland, Oregon: Fish Commission of Oregon.

Schoonover, J.E., and K.W.J. Williard. 2003. Groundwater Nitrate Reduction in Giant Cane and Forest Riparian Buffer Zones. *Journal of the American Water Resources Association* 39(2): 347-354.

Schuett-Hames, D., B. Conrad, A. Pleus, and K. Lautz. 1996. Literature Review and Monitoring Recommendations for Salmonid Spawning Gravel Scour. TFW-AM9-96-001. Northwest Indian Fisheries Commission and Washington Department of Fish and Wildlife.

Schwarz, A.L., and G.L. Greer. 1984. Responses of Pacific Herring, *Clupea Barengus* Pallast, to Some Underwater Sounds. *Canadian Journal of Fisheries and Aquatic Sciences* 41: 1183-1192.

Sear, D.A. 1995. Morphological and Sediment logical Changes in a Gravel-Bed River Following 12 Years of Flow Regulation for Hydropower. *Regulated Rivers: Research and Management* 10: 247-264.

Sedell, J.R., F.J. Swanson, and S.V. Gregory. 1985. Evaluating Fish Response to Woody Debris. In *Proceedings of the Pacific Northwest Stream Habitat Workshop*, edited by T.J. Hassler. Arcata, California: California Cooperative Fishery Research Unit, Humboldt State University. pp. 222-245

Seitz, R.D., R.N. Lipcius, and M.S. Seebo. 2005. Food Availability and Growth of the Blue Crab in Seagrass and Unvegetated Nurseries of Chesapeake Bay. *Journal of Experimental Marine Biology and Ecology* 319(1-2): 57-68.

Servizi, J.A., and D.W. Martens. 1987. Some Effects of Suspended Fraser River Sediments on Sockeye Salmon. *Canadian Special Publication of Fisheries and Aquatic Sciences* 96: 254-264.

Servizi, J.A., and D.W. Martens. 1991. Effect of Temperature, Season, and Fish Size on Acute Lethality of Suspended Sediments to Coho Salmon (*Oncorhynchus-Kisutch*). *Canadian Journal of Fisheries and Aquatic Sciences* 48(3): 493-497.

Servizi, J.A., and D.W. Martens. 1992. Sublethal Responses of Coho Salmon (*Oncorhynchus-Kisutch*) to Suspended Sediments. *Canadian Journal of Fisheries and Aquatic Sciences* 49(7): 1389-1395.

Servizi, J.A., and R.W. Gordon. 1990. Acute Lethal Toxicity of Ammonia and Suspended Sediment Mixtures to Chinook Salmon (*Oncorhynchus Tshawytscha*). *Bulletin of Environmental Contamination and Toxicology* 44(4): 650-656.

Shaffer, J.A., D. Penttila, M. McHenry, and D. Vilella. 2007. Observations of Eulachon, *Thaleichthys Pacificus*, in the Elwha River, Olympic Peninsula, Washington. *Northwest Science* 81(1): 76-81.

Sharber, N.G., and S.W. Carothers. 1988. Influence of Electrofishing Pulse Shape on Spinal Injuries in Adult Rainbow Trout. *North American Journal of Fisheries Management* 8: 117-122.

Shaw, T.C. 1996. Effectiveness of an Excluder Device in Preventing the Entrainment of Benthic Invertebrates and Demersal Fishes in Grays Harbor, Washington. Proceedings of the Western Dredging Association Seventeenth Technical Conference.

Sheibley, R.W., A.P. Jackman, J.H. Duff, and F.J. Triska. 2003. Numerical Modeling of Coupled Nitrification-Denitrification in Sediment Perfusion Cores from the Hyporheic Zone of the Shingobee River, Minnesota. *Advances in Water Resources* 26(9): 977-987.

Sheibley, R.W., D.S. Ahearn, and R.A. Dahlgren. 2006. Nitrate Loss from a Restored Floodplain in the Lower Cosumnes River, California. *Hydrobiologia* 571: 261-272.

Sheibley, R.W., J.H. Duff, A.P. Jackman, and F.J. Triska. 2003. Inorganic Nitrogen Transformations in the Bed of the Shingobee River, Minnesota: Integrating Hydrologic and Biological Processes Using Sediment Perfusion Cores. *Limnology and Oceanography* 48: 1129-1140.

Shepard, B.B., K.L. Pratt, and P.J. Graham. 1984. Life Histories of Westslope Cutthroat Trout and Bull Trout in the Upper Flathead River Basin, Montana. Helena, Montana: Montana Department of Fish, Wildlife and Parks.

Sherwood, C.R., D.A. Jay, R.B. Harvey, P. Hamilton, and C.A. Simenstad. 1990. Historical Changes in the Columbia River Estuary. *Progress in Oceanography* 25(1-4): 299-352.

Shields, F.D. 1991. Woody Vegetation and Riprap Stability Along the Sacramento River Mile 84.5-119. *Water Resources Bulletin* 27(3): 527-536.

Shields, F.D., and D.H. Gray. 1992. Effects of Woody Vegetation on Levee Integrity *Water Resources Bulletin* 28: 917-931.

Shuman, J.R. 1995. Environmental Considerations for Assessing Dam Removal Alternatives for River Restoration. *Regulated Rivers-Research & Management* 11(3-4): 249-261.

Sigler, J.W., T. Bjornn, and E.H. Everest. 1984. Effects of Chronic Turbidity on Density and Growth of Steelheads and Coho Salmon. *Transaction of the American Fisheries Society* 113: 142-150.

Simenstad, C.A., B. Nightingale, R.A. Thom, and D.K. Shreffler. 1999. Impacts of Ferry Terminals on Juvenile Salmon Migrating Along Puget Sound Shorelines: Phase I Synthesis of State of Knowledge. Research Project T9903 Task A2. Seattle, Washington: Washington State Transportation Center.

Simenstad, C.A., B.S. Miller, C.F. Nyblade, K. Thornburgh, and L.J. Bledsoe. 1979. Food Web Relationship of Northern Puget Sound and the Strait of Juan de Fuca. EPA Interagency Agreement No. D6-E693-EN. Office of Environmental Engineering and Technology, U.S. Environmental Protection Agency.

Simenstad, C.A., W.J. Kinney, S.S. Parker, E.O. Salo, J.R. Cordell, and H. Buechner. 1980. Prey Community Structure and Trophic Ecology of Outmigrating Juvenile Chum and Pink Salmon in Hood Canal, Washington: A Synthesis of Three Years' Studies, 1977-1979. Seattle, Washington: University of Washington.

Simon, A. 1994. Gradation Processes and Channel Evolution in Modified West Tennessee Streams Process, Response, and Form. Professional Paper 1470, Denver, Colorado: U.S. Geological Survey, 84p.

Simon, A., and C.R. Hupp. 1992. Geomorphic and Vegetative Recovery Processes Along Modified Tennessee Streams: An Interdisciplinary Approach to Disturbed Fluvial Systems. Washington D.C. International Association of Hydrological Sciences.

Sinclair, M. 1992. *Marine Populations - an Essay on Population Regulation and Speciation*. Books in Recruitment Fishery Oceanography. Seattle and London: Washington Sea Grant Program, University of Washington Press.

Snoeyink, V.L., and D. Jenkins. 1980. *Water Chemistry*. New York: Wiley.

Snyder, D.E. 2003. Electrofishing and Its Harmful Effects on Fish. Information and Technology Report USGS/BRD/ITR—2003-0002. Denver, Colorado: U.S. Geological Survey.

Sobocinski, K.L. 2003. The Impact of Shoreline on Supratidal Beach Fauna of Central Puget Sound. Seattle, Washington: University of Washington, 83 pp.

Sommer, T.R., M.L. Nobriga, W.C. Harrell, W. Batham, and W.J. Kimmerer. 2001. Floodplain Rearing of Juvenile Chinook Salmon: Evidence of Enhanced Growth and Survival. *Canadian Journal of Fisheries and Aquatic Sciences* 58(2): 325-333.

Spanhoff, B., W. Riss, P. Jakel, N. Dakkak, and E.I. Meyer. 2006. Effects of an Experimental Enrichment of Instream Habitat Heterogeneity on the Stream Bed Morphology and Chironomid Community of a Straightened Section in a Sandy Lowland Stream. *Environmental Management* 37(2): 247-257.

Spence, B.C., G.A. Lomnicky, R.M. Hughes, and R.P. Novitzki. 1996. An Ecosystem Approach to Salmonid Conservation. TR-4501-96-6057. Corvallis, Oregon: ManTech Environmental Research Services Corporation.

Sridhar, V., A.L. Sansone, J. LaMarche, T. Dubin, and D.P. Lettenmaier. 2004. Prediction of Stream Temperature in Forested Watersheds. *Journal of the American Water Resources Association* 40(1): 197-213.

Stadler, J., NOAA Fisheries, Seattle, Washington. 2007. Email regarding NOAA Fisheries use of the Practical Spreading Loss model to estimate underwater noise intensity for the purpose of ESA consultation; sent to Eric Doyle of Herrera Environmental Consultants, Inc. Seattle, Washington, August 29, 2007.

Stanford, J.A., and J.V. Ward. 1992. Management of Aquatic Resources in Large Catchments: Recognizing Interactions between Ecosystem Connectivity and Environmental Disturbance. In *Watershed Management – Balancing Sustainability and Environmental Change*, edited by R.J. Naiman. New York: Springer-Verlag.

Stanford, J.A., and J.V. Ward. 1993. An Ecosystem Perspective of Alluvial Rivers - Connectivity and the Hyporheic Corridor. *Journal of the North American Benthological Society* 12(1): 48-60.

Stanford, J.A., J.V. Ward, W.J. Liss, C.A. Frissell, R.N. Williams, J.A. Lichatowich, and C.C. Coutant. 1996. A General Protocol for Restoration of Regulated Rivers. *Regulated Rivers: Research & Management* 12(4-5): 391-413.

Stanley, D.R., and C.A. Wilson. 2004. Effect of Hypoxia on the Distribution of Fishes Associated with a Petroleum Platform Off Coastal Louisiana. *North American Journal of Fisheries Management* 24(2): 662-671.

Starr, R.M., D.S. Fox, M.A. Hixon, B.N. Tissot, G.E. Johnson, and W.H. Barss. 1996. Comparison of Submersible-Survey and Hydroacoustic Survey Estimates of Fish Density on a Rocky Bank. *Fishery Bulletin* 94: 113-123.

Stein, D., and T.J. Hassler. 1989. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Southwest): Brown Rockfish, Copper Rockfish, and Black Rockfish. U.S. Fish and Wildlife Service.

Stein, J.E., T. Hom, T.K. Collier, D.W. Brown, and U. Varanasi. 1995. Contaminant Exposure and Biochemical Effects in Outmigrant Juvenile Chinook Salmon from Urban and Nonurban Estuaries of Puget-Sound, Washington. *Environmental Toxicology and Chemistry* 14(6): 1019-1029.

Stocker, M. 2002. Fish Mollusks and Other Sea Animals, and the Impact of Anthropogenic Noise in the Marine Acoustical Environment. Michael Stocker Associates for Earth Island Institute.

Stone, J., and S. Barndt. 2005. Spatial Distribution and Habitat Use of Pacific Lamprey (*Lampetra Tridentata*) Ammocoetes in a Western Washington Stream. *Journal of Freshwater Ecology* 20(1): 171-185.

Sullivan, K., D.J. Martin, R.D. Cardwell, J.E. Toll, and S. Duke. 2000. *An Analysis of the Effects of Temperature on Salmonids of the Pacific Northwest with Implications for Selecting Temperature Criteria*. Portland, Oregon: Sustainable Ecosystems Institute.

Sullivan, K., J. Tooley, K. Doughty, J.E. Caldwell, and P. Knudsen. 1990. Evaluation of Prediction Models and Characterization of Stream Temperature Regimes in Washington. TFW-WQ3-90-006. Olympia, Washington: Washington Department of Natural Resources.

Sumida, B.Y., and H.G. Moser. 1984. Food and Feeding of Bocaccio and Comparison with Pacific Hake Larvae in the California Current. Rept. 25:112-118. La Jolla, California: *California Cooperative Oceanic Fisheries Investigations*.

Suttle, K.B., M.E. Power, J.M. Levine, and C. McNeely. 2004. How Fine Sediment in Riverbeds Impairs Growth and Survival of Juvenile Salmonids. *Ecological Applications* 14(4): 969-974.

Takami, T., F. Kitano, and S. Nakano. 1997. High Water Temperature Influences on Foraging Responses and Thermal Deaths of Dolly Varden *Salvelinus Malma* and White-Spotted Charr *S-Leucomaenis* in a Laboratory. *Fisheries Science* 63(1): 6-8.

Teachout, E., Fish and Wildlife Biologist with U.S. Fish and Wildlife Service, Lacey, Washington. 2007. Email regarding USFWS use of the Practical Spreading Loss model to estimate underwater noise intensity for the purpose of ESA consultation; sent to Julie Hampden of Herrera Environmental Consultants, Inc. Seattle, Washington, August 29, 2007.

Teodoru, C., and B. Wehrli. 2005. Retention of Sediments and Nutrients in the Iron Gate I Reservoir on the Danube River. *Biogeochemistry* 76(3): 539-565.

Terich, T.A. 1987. *Living with the Shore of Puget Sound and the Georgia Strait*. Living with the Shore. Edited by O.H. Pilkey and W.J. Neal. Durham, NC: Duke University Press.

Terrados, J., C.M. Duarte, M.D. Fortes, J. Borum, N.S.R. Agawin, S. Bach, U. Thampanya, L. Kamp-Nielsen, W.J. Kenworthy, O. Geertz-Hansen, and J. Vermaat. 1998. Changes in Community Structure and Biomass of Seagrass Communities Along Gradients of Siltation in SE Asia. *Estuarine Coastal and Shelf Science* 46(5): 757-768.

Theurer, F.D., K.A. Voos, and W.J. Miller. 1984. Instream Water Temperature Model. Instream Flow Informational Paper 16 FWS/OBS-84/15. U.S. Fish and Wildlife Services.

Thom, R.M., and D.K. Shreffler. 1996. Eelgrass Meadows near Ferry Terminals in Puget Sound. Sequim, Washington: Battelle Pacific Northwest Laboratories.

Thom, R.M., C.A. Simenstad, J.R. Cordell, and E.O. Salo. 1989. Fish and Their Epibenthic Prey in a Marina and Adjacent Mudflats and Eelgrass Meadow in a Small Estuarine Bay. Seattle, Washington: University of Washington.

Thom, R.M., D.K. Shreffler, and K. Macdonald. 1994. Shoreline Armoring Effects on Coastal Ecology and Biological Resources in Puget Sound, Washington. Shorelands and Environmental Assistance Program, Washington Department of Ecology.

Thompson, K.G., E.P. Bergersen, R.B. Nehring, and D.C. Bowden. 1997. Long-Term Effects of Electrofishing on Growth and Body Condition of Brown and Rainbow Trout. *North American Journal of Fisheries Management* 17: 154-159.

Thoms, M.C. 2003. Floodplain-River Ecosystems: Lateral Connections and the Implications of Human Interference. *Geomorphology* 56(3-4): 335-349.

Thomson Scientific Web of Science. 2007. Available at: http://scientific.thomson.com/products/wos/ (accessed May–July 2007).

Thorkilsen, M., and C. Dynesen. 2001. An Owner’s View of Hydroinformatics: It’s Role in Realizing the Bridge and Tunnel Connection between Denmark and Sweden. *Journal of Hydroinformatics* 3(2): 105-135.

Tiemann, J.S., D.P. Gillette, M.L. Wildhaber, and D.R. Edds. 2004. Effects of Lowhead Dams on Riffle-Dwelling Fishes and Macroinvertebrates in a Midwestern River. *Transactions of the American Fisheries Society* 133(3): 705-717.

Tiemann, J.S., H.R. Dodd, N. Owens, and D.H. Wahl. 2007. Effects of Lowhead Dams on Unionids in the Fox River, Illinois. *Northeastern Naturalist* 14(1): 125-138.

Tockner, K., D. Pennetzdorfer, N. Reiner, F. Schiemer, and J.V. Ward. 1999. Hydrological Connectivity, and the Exchange of Organic Matter and Nutrients in a Dynamic River-Floodplain System (Danube, Austria). *Freshwater Biology* 41(3): 521-535.

Toft, J., C. Simenstad, J. Cordell, and L. Stamatiou. 2004. Fish Distribution, Abundance, and Behavior at Nearshore Habitats Along City of Seattle Marine Shorelines, with an Emphasis on Juvenile Salmonids. Prepared for Seattle Public Utilities by Wetland Ecosystem Team, University of Washington School of Aquatic and Fishery Sciences, Seattle, Washington. March 2004.

Tops, S., W. Lockwood, and B. Okamura. 2006. Temperature-Driven Proliferation of *Tetracapsuloides Bryosalmonae* in Bryozoan Hosts Portends Salmonid Declines. *Diseases of Aquatic Organisms* 70(3): 227-236.

Travnichek, V.H., A.V. Zale, and W.L. Fisher. 1993. Entrainment of Ichthyoplankton by a Warmwater Hydroelectric Facility. *Transactions of the American Fisheries Society* 122: 709-716.

Travnichek, V.H., M.B. Bain, and M.J. Maceina. 1995. Recovery of a Warmwater Fish Assemblage after the Initiation of a Minimum Flow-Release Downstream from a Hydroelectric Dam. *Transactions of the American Fisheries Society* 124: 836-844.

Triska, F.J., V.C. Kennedy, R.J. Avanzino, G.W. Zellweger, and K.E. Bencala. 1989. Retention and Transport of Nutrients in a 3rd-Order Stream in Northwestern California - Hyporheic Processes. *Ecology* 70: 1893-1905.

Turnpenny, A.W.H., K.P. Thatcher, and J.R. Nedwell. 1994. The Effects on Fish and Other Marine Animals of High-Level Underwater Sound. Fawley Aquatic Research.

Urban, E.R., and C.J. Langdon. 1984. Reduction in Costs of Diets for the American Oyster, *Crassostrea-Virginica* (Gmelin), by the Use of Non-Algal Supplements. *Aquaculture* 38(4): 277-291.

Urban, E.R., and D.L. Kirchman. 1992. Effect of Kaolinite Clay on the Feeding-Activity of the Eastern Oyster *Crassostrea-Virginica* (Gmelin). *Journal of Experimental Marine Biology and Ecology* 160(1): 47-60.

Urick, R.J. 1983. Principles of Underwater Sound. Ch. 7 In *The Noise Background of the Sea*, Los Altos, California: Peninsula Publishing.

USACE. 2007. Dredging Operations and Environmental Research Program. March 2007. Available at U.S. Army Corps of Engineers website, URL = http://el.erdc.usace.army.mil/dots/doer/ (accessed September 3, 2007).

USEPA. 2007. National Management Measures to Control Nonpoint Source Pollution from Hydromodification. EPA 841-B-07-002. Washington, DC: U.S. Environmental Protection Agency, Office of Water.

USFS, NMFS, USBLM, USFWS, USNPS, and USEPA. 1993. Forest Ecosystem Management: An Ecological, Economic, and Social Assessment. Portland, Oregon: USFS PNW Region.

USFS. 2007. Fish Resources. Available at agency website: http://www.fs.fed.us/r6/fishing/forests/fishresources/win_coldwater.html#redband).

USFWS. 1986. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest) Steelhead Trout. Biological Report 82 (11.82). Lafayette, Louisiana: U.S. Fish and Wildlife Service, Coastal Ecology Group.

Vagle, S. 2003. On the Impact of Underwater Pile Driving Noise on Marine Life. Government Report. Canada Department of Fisheries and Oceans, Institute of Ocean Sciences, Ocean Science and Productivity Division.

Valett, H.M., C.L. Crenshaw, and P.F. Wagner. 2002. Stream Nutrient Uptake, Forest Succession, and Biogeochemical Theory. *Ecology* 83(10): 2888-2901.

Valett, H.M., M.A. Baker, J.A. Morrice, C.S. Crawford, M.C. Molles, C.N. Dahm, D.L. Moyer, J.R. Thibault, and L.M. Ellis. 2005. Biogeochemical and Metabolic Responses to the Flood Pulse in a Semiarid Floodplain. *Ecology* 86(1): 220-234.

Vallania, A., and M.D. Corigliano. 2007. The Effect of Regulation Caused by a Dam on the Distribution of the Functional Feeding Groups of the Benthos in the Sub Basin of the Grande River (San Luis, Argentina). *Environmental Monitoring and Assessment* 124(1-3): 201-209.

Van Eenennaam, J.P., J. Linares-Casenave, X. Deng, and S.I. Doroshov. 2005. Effect of Incubation Temperature on Green Sturgeon Embryos, *Acipenser Medirostris*. *Environmental Biology of Fishes* 72(2): 145-154.

Vandenavyle, M.J., and M.A. Maynard. 1994. Effects of Saltwater Intrusion and Flow Diversion on Reproductive Success of Striped Bass in the Savanna River Estuary. *Transactions of the American Fisheries Society* 123(6): 886-903.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Science* 37: 130-137.

Vaudo, J.J., and C.G. Lowe. 2006. Movement Patterns of the Round Stingray *Urobatis Halleri* (Cooper) near a Thermal Outfall. *Journal of Fish Biology* 68(6): 1756-1766.

Vellidis, G., R. Lowrance, P. Gay, and R.K. Hubbard. 2003. Nutrient Transport in a Restored Riparian Wetland. *Journal of Environmental Quality* 32(2): 711-726.

Vogel, J.L., and D.A. Beauchamp. 1999. Effects of Light, Prey Size, and Turbidity on Reaction Distances of Lake Trout (*Salvelinus Namaycush*) to Salmonid Prey. *Canadian Journal of Fisheries and Aquatic Sciences* 56(7): 1293-1297.

WAC 173-201A. 2006. Water Quality Standards for Surface Waters of the State of Washington. Washington Administrative Code.

Wagner, C.M., and H.M. Austin. 1999. Correspondence between Environmental Gradients and Summer Littoral Fish Assemblages in Low Salinity Reaches of the Chesapeake Bay, USA. *Marine Ecology-Progress Series* 177: 197-212.

Wagner, E.J., R.E. Arndt, and M. Brough. 2001. Comparative Tolerance of Four Stocks of Cutthroat Trout to Extremes in Temperature, Salinity, and Hypoxia. *Western North American Naturalist* 61(4): 434-444.

Wagner, E.J., T. Bosakowski, and S. Intelmann. 1997. Combined Effects of Temperature and High pH on Mortality and the Stress Response of Rainbow Trout after Stocking. *Transactions of the American Fisheries Society* 126(6): 985-998.

Wantzen, K.M. 2006. Physical Pollution: Effects of Gully Erosion on Benthic Invertebrates in a Tropical Clear-Water Stream. *Aquatic Conservation-Marine and Freshwater Ecosystems* 16(7): 733-749.

Warmke, S., and D. Hering. 2000. Composition, Microdistribution and Food of the Macroinvertebrate Fauna Inhabiting Wood in Low-Order Mountain Streams in Central Europe. *International Review of Hydrobiology* 85(1): 67-78.

Wasson, K., K. Fenn, and J.S. Pearse. 2005. Habitat Differences in Marine Invasions of Central California. *Biological Invasions* 7(6): 935-948.

Waters, T.F. 1995. Sediment in Streams: Sources, Biological Effects and Control. Bethesda, Maryland: American Fisheries Society.

Watson, L.R., A. Milani, and R.P. Hedrick. 1998. Effects of Water Temperature on Experimentally-Induced Infections of Juvenile White Sturgeon (*Acipenser Transmontanus*) with the White Sturgeon Iridovirus (WSIV). *Aquaculture* 166(3-4): 213-228.

Watters, G.T. 1999. Freshwater Mussels and Water Quality: A Review of the Effects of Hydrologic and Instream Habitat Alterations. *First Freshwater Mollusk Conservation Society Symposium*, Columbus, Ohio.

WDFW. 1997a. Washington State Forage Fish: Sand Lance Webpage. Available at Washington Department of Fish and Wildlife website: http://wdfw.wa.gov/fish/forage/lance.htm (accessed July 2007).

WDFW. 1997b. Washington State Forage Fish Fact Sheet: Puget Sound Herring Fact Sheet. Available at Washington Department of Fish and Wildlife website: http://www.wdfw.wa.gov/fish/forage/herring.htm (accessed November 3, 2006).

WDFW. 1997c. Washington State Forage Fish Fact Sheet: Washington State Surf Smelt Fact Sheet. Available at Washington Department of Fish and Wildlife website: http://www.wdfw.wa.gov/fish/forage/smelt.htm (accessed November 3, 2006).

WDFW. 2001. Washington and Oregon Eulachon Management Plan. Olympia, Washington: Washington Department of Fish and Wildlife.

WDFW. 2003. Integrated Streambank Protection Guidelines. Olympia, Washington: Washington Department of Fish and Wildlife.

WDNR. 2005. Forest Practices - Habitat Conservation Plan. Olympia, Washington: Washington State Department of Natural Resources.

WDNR. 2006a. Draft Covered Species Paper - Fish. Olympia, Washington: Washington Department of Natural Resources.

WDNR. 2006b. Draft Covered Species Paper - Invertebrates. Olympia, Washington: Washington Department of Natural Resources.

Webber, J.D., S.N. Chun, T.R. MacColl, L.T. Mirise, A. Kawabata, E.K. Anderson, T.S. Cheong, L. Kavvas, M.M. Rotondo, K.L. Hochgraf, R. Churchwell, and J.J. Cech. 2007. Upstream Swimming Performance of Adult White Sturgeon: Effects of Partial Baffles and a Ramp. *Transactions of the American Fisheries Society* 136(2): 402-408.

Wedemeyer, G.A., R.L. Saunders, and W.C. Clarke. 1980. Environmental-Factors Affecting Smoltification and Early Marine Survival of Anadromous Salmonids. *Marine Fisheries Review* 42(6): 1-14.

Weitkamp, D.E., and T.H. Schadt. 1982. 1980 Juvenile Salmonid Study. Prepared for the Port of Seattle, Washington: Prepared by Parametrix, Inc.

Welch, E.B., J.M. Jacoby, and C.W. May. 1998. Stream Quality. In *River Ecology and Management*, edited by R.J. Naiman and R.E. Bilby. New York: Springer. pp. 69-94.

Welker, T.L., S.T. McNulty, and P.H. Klesius. 2007. Effect of Sublethal Hypoxia on the Immune Response and Susceptibility of Channel Catfish, *Ictalurus Punctatus*, to Enteric Septicemia. *Journal of the World Aquaculture Society* 38(1): 12-23.

Wenning, R.J., D.B. Mathur, D.J. Paustenbach, M.J. Stephenson, S. Folwarkow, and W.J. Luksemburg. 1999. Polychlorinated Dibenzo-P-Dioxins and Dibenzofurans in Storm Water Outfalls Adjacent to Urban Areas and Petroleum Refineries in San Francisco Bay, California. *Archives of Environmental Contamination and Toxicology* 37(3): 290-302.

West, J. 1997. Protection and Restoration of Marine Life in the Inland Waters of Washington State. Puget Sound/Georgia Basin Environmental Report Series (23):Number 6. Olympia, Washington: Puget Sound Water Quality Action Team.

West, J.E., R.M. Buckley, and D.C. Doty. 1994. Ecology and Habitat Use of Juvenile Rockfishes (*Sebastes* Spp.) Associated with Artificial Reefs in Puget-Sound, Washington. *Bulletin of Marine Science* 55(2-3): 344-350.

White, D.S. 1990. Biological Relationships to Convective Flow Patterns within Stream Beds *Hydrobiologia* 196: 149-158.

Widdows, J., P. Fieth, and C.M. Worrall. 1979. Relationships between Seston, Available Food and Feeding-Activity in the Common Mussel *Mytilus-Edulis*. *Marine Biology* 50(3): 195-207.

Wik, S.J. 1995. Reservoir Drawdown: Case Study in Flow Changes to Potentially Improve Fisheries. *Journal of Energy Engineering* 121(2): 89-96.

Wilber, D.H., and D.G. Clarke. 2001. Biological Effects of Suspended Sediments: A Review of Suspended Sediment Impacts on Fish and Shellfish with Relation to Dredging Activities in Estuaries. *North American Journal of Fisheries Management* 21(4): 855-875.

Wildish, D.J., and J. Power. 1985. Avoidance of Suspended Sediments by Smelt as Determined by a New “Single Fish” Behavioral Bioassay. *Bulletin of Environmental Contamination and Toxicology* 34: 770-774.3.

Williams, G.D., and R.M. Thom. 2001. Marine and Estuarine Shoreline Modification Issues White Paper. Olympia, Washington: Washington Department of Fish and Wildlife.

Williams, G.D., R.M. Thom, and J.E. Starkes. 2001. Reconnaissance Assessment of the State of the Nearshore Ecosystem: Eastern Shore of Central Puget Sound, Including Vashon and Maury Islands (WRIAs 8 and 9). Prepared for King County Department of Natural Resources, Seattle, Washington.

Williams, J.D., M.L. Warren, K.S. Cummings, J.L. Harris, and R.J. Neves. 1993. Conservation Status of Fresh-Water Mussels of the United-States and Canada. *Fisheries* 18(9): 6-22.

Willson, M.F., R.H. Armstrong, M.C. Hermans, and K. Koski. 2006. Eulachon: A Review of Biology and an Annotated Bibliography. Alaska Fisheries Science Center and NOAA Fisheries.

Wilzbach, M.A., B.C. Harvey, J.L. White, and R.J. Nakamoto. 2005. Effects of Riparian Canopy Opening and Salmon Carcass Addition on the Abundance and Growth of Resident Salmonids. *Canadian Journal of Fisheries and Aquatic Sciences* 62(1): 58-67.

Winger, P.V., and P.J. Lasier. 1994. Effects of Salinity on Striped Bass Eggs and Larvae from the Savanna River, Georgia. *Transactions of the American Fisheries Society* 123(6): 904-912.

Winn, R.N., and D.M. Knott. 1992. An Evaluation of the Survival of Experimental Populations Exposed to Hypoxia in the Savanna River Estuary. *Marine Ecology-Progress Series* 88(2-3): 161-179.

Winter, B.D. 1990. A Brief Review of Dam Removal Efforts in Washington, Oregon, Idaho, and California. Technical Memo NMFS F/NWR-28. U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

Winter, H.V., and W.L.T. Van Densen. 2001. Assessing the Opportunities for Upstream Migration of Non-Salmonid Fishes in the Weir-Regulated River Vecht. *Fisheries Management and Ecology* 8(6): 513-532.

Wisby, W.J., J.D. Richard, D.R. Nelson, and S.H. Gruber. 1964. Sound Perception in Elasmobranches. In *Marine Bio-Acoustics*, edited by W.N. Tavolga. New York: Pergamon Press. pp. 255-268.

Wohl, E.E. 2004. *Disconnected Rivers: Linking Rivers to Landscapes*. New Haven: Yale University Press.

Wood, P.J., and P.D. Armitage. 1997. Biological Effects of Fine Sediment in the Lotic Environment. *Environmental Management* 21: 203-217.

WSDOT. 2006. Biological Assessment Preparation for Transportation Projects, Advanced Training Manual. Olympia, Washington: Washington State Department of Transportation.

Wydoski, R.G., and R.S. Wydoski. 2002. Age, Growth, and Reproduction of Mountain Suckers in Lost Creek Reservoir, Utah. *Transactions of the American Fisheries Society* 131(2): 320-328.

Wydoski, R.S., and R.R. Whitney. 2003. *Inland Fishes of Washington, Second Edition*. American Fisheries Society and University of Washington Press.

Young, W.T., and D.L. Scarnecchia. 2005. Habitat Use of Juvenile White Sturgeon in the Kootenai River, Idaho and British Columbia. *Hydrobiologia* 537: 265-271.

Zemlak, R.J., and J.D. McPhail. 2006. The Biology of Pygmy Whitefish, *Prosopium Coulterii*, in a Closed Sub-Boreal Lake: Spatial Distribution and Diel Movements. *Environmental Biology of Fishes* 76(2-4): 317-327.

Zhang, W., X.L. Chen, J.H. Xu, G.C. Li, and Z.Q. Wang. 2000. Experimental Study on Scours Downstream of Floodgates. *China Ocean Engineering* 14(2): 243-254.

Zimmermann, A.E., and M. Lapointe. 2005. Intergranular Flow Velocity through Salmonid Redds: Sensitivity to Fines Infiltration from Low Intensity Sediment Transport Events. *River Research and Applications* 21(8): 865-881.